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ESTUARINE COMMUNITY DYNAMICS IN RELATION  
TO POWER PLANT OPERATIONS

Benthic Process Program  
Final Report

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Co-Principal Investigators

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## ABSTRACT

This study had two major objectives. The first was to conduct in-situ studies of major carbon, nitrogen, phosphorus and sediment rate processes such as sedimentation, resuspension, benthic fluxes of nutrients and organic matter and water column metabolism. The second objective was to develop budgets of nitrogen, phosphorus and carbon in 3 distinctive areas of the estuary, including the area potentially influenced by power plant operations.

To meet these objectives, stations were located in the near-field at the terminus of the discharge canal and in the far-field near Buena Vista, approximately 6 km downstream. Because there are strong longitudinal gradients in water quality variables, two additional stations were selected, one above the power plant at Jones Point and one downstream of the plant at Marsh Point. The two stations in the vicinity of the plant were in the high turbidity, low salinity zone while the upstream and downstream stations were in the tidal-fresh and moderately stratified mesohaline zones, respectively. Thus, the distribution of sampling stations reflected 3 distinctive regions of the estuary and included a site in the immediate vicinity of the power plant.

At these primary locations, estimates were made of water column metabolism, benthic respiration, benthic nutrient fluxes, water quality characteristics of the sediments and overlying waters, light extinction and sedimentation and resuspension rates of particulate materials. On a quarterly basis, water quality surveys were conducted along the mainstem of the estuary with 10 stations arranged from the mouth to river kilometer 72. Four, 13-hour tidal surveys were also conducted at 3 different stations to estimate tidally-induced variability in water quality variables collected during synoptic longitudinal surveys. Finally, 4 water quality surveys were conducted in the intake-discharge canals at the power plant and were designed to allow estimates to be made of net gains and losses of carbon, nitrogen and phosphorus related to power plant operation. These measurements were made in 1980-1981.

This report contains major sections which deal with nutrient processes, organic matter dynamics, section, sediment/water budgets for nitrogen, phosphorus and carbon, and an evaluation of power plant induced perturbations. A methodology chapter, covering all topics, precedes the major sections.

#### ACKNOWLEDGMENTS

We are indebted to a number of people whose assistance greatly enhanced the conduct of this work. Mark Jenkins, Linda Brieck, and Janet Barnes provided enthusiastic assistance during long and tedious field surveys. The helpful and able seamanship of Captain William C. Keefe and Mate John Crane contributed to the completion of this work. We thank Dr. Frederick Holland, contract coordinator (Martin Marietta Corporation) for his assistance in both technical and administrative matters and Dr. Randy Roig, Maryland Power Plant Siting Program, for his patient understanding and enthusiasm for our work. Mr. William Caplins assisted in statistical analyses. Computational support was provided, in part, by the University of Maryland Computer Science Center. We also thank Fran Younger who drafted the figures and Betty Ashby who typed the manuscript.

## INTRODUCTION

Estuaries are characterized by relatively high rates of primary and secondary production when compared to many coastal and oceanic systems. We have recently completed a review of 63 estuarine systems spanning a spectrum from highly stratified to vertically-mixed and from river-dominated to lagoonal and found numerous examples in each category where average annual rates exceeded  $0.5 \text{ g C m}^{-2} \text{ d}^{-1}$  and maximal rates often approached or exceeded  $4 \text{ g C m}^{-2} \text{ d}^{-1}$  (Boynton, et al. 1982). There are numerous mechanisms that have been proposed as being more or less responsible for sustaining these high rates. Among them are the notions that (1) estuaries receive large amounts of essential nutrients from riverine inputs, (2) the two-layered circulation prevailing in some estuaries tends to conserve nutrients which are essential in sustaining high productivity rates, (3) in-welling from deeper and nutrient-rich coastal waters constitutes a significant nutrient source, (4) marshes and other wetlands associated with some estuarine systems export nutrients which can be utilized by phytoplankton, and (5) most estuaries are relatively shallow and, because of this, zones of production and remineralization (both water column and benthic) are closely coupled in space (Nixon, 1981).

While a comprehensive synthesis on estuarine primary production-nutrient patterns and strategies is still lacking, quantitative data on major pathways continues to be collected. As Nixon (1979) has pointed out, our perspective on the importance of various components in the production-nutrient cycling scheme has changed considerably over the last few years. One of these changes

has been the documentation of benthic remineralization as an important source of recycled nutrients available for phytoplankton photosynthesis (Davies 1975; Rowe et al., 1977; Boynton et al., 1980a). This is in contrast to the prevalent view that water-column processes dominate estuarine remineralization (e.g. McCarthy et al. 1974; Carpenter and McCarthy 1978). It is of course tempting to argue the extremes of each case, but intuitively it seems more reasonable to assume that the relative importance of these remineralization sites changes on a continuum from water-column dominated to benthic dominated.

Whatever the actual mixture of mechanisms responsible for nutrient cycling, the coupling of nutrients to primary production is of central importance in most aquatic ecosystems because of its role in supporting many, if not most, food webs. Strickland (1961) emphasized this by saying that disinterest in algal production would "not be unlike the situation which would prevail if livestock raisers lacked knowledge of, or interest in, pasture conditions."

#### Patuxent River Studies

One of the most extensively studied estuaries on the east coast of the United States is that of the Patuxent River, a tributary of Chesapeake Bay. Measurements of water column nutrient concentrations, which were initiated prior to the 1940's (e.g. Newcomb and Brust 1940; Nash 1947), have continued into the 1980's (e.g. Cory and Nauman 1967; Flemer et al. 1970, Sage, pers. comm.). A few estimates of nutrient fluxes have been made for watershed inputs from point (FWPCA 1968) and diffuse (Correll 1977) sources and for exchanges with brackish marshes (Heinle 1976), while net biochemical fluxes

have been estimated as the residual term in a one-dimensional model of longitudinal mass-transport along the estuary (Ulanowicz and Flemer 1977). For the past several years, we have been conducting nutrient studies in the Patuxent estuary both at the scale of the entire watershed for purposes of constructing nutrient budgets (Kemp and Boynton 1979) and at a distinctive location in the estuary (turbidity maximum zone) to investigate nutrient cycling between the water column and the benthos and possible interactions of a steam electric generation station with these processes. Results of this work (Boynton et al. 1980a) indicated that benthic respiration rates ranged from 0.5 to 4.1 g O<sub>2</sub> m<sup>-2</sup> d<sup>-1</sup> and were positively correlated with temperature and primary production. Net fluxes of ammonium (NH<sub>4</sub><sup>+</sup>) and dissolved inorganic phosphorus (DIP) ranged from -105 to 1584 µg-at N m<sup>-2</sup> h<sup>-1</sup> and 1 to 295 µg-at P m<sup>-2</sup> h<sup>-1</sup>, respectively. These rates, which were positively correlated with temperature, are among the highest yet reported in the literature. Fluxes of nitrate plus nitrite were small during summer when the water column concentrations were low, but high and directed into the sediments during winter when water column concentrations were high. In general, it appears that nutrient fluxes across the sediment-water interface represent an important source to the water column when photosynthetic demand is high and water column stocks are low and, conversely, serve as a sink in winter when demand is low and water column stocks high, thereby serving a "buffering" function between supply and demand. A simple budget of sediment-water exchanges and storages of nitrogen indicated that, of the total particulate nitrogen deposited annually onto the sediments, about 34% was returned to the water column as NH<sub>4</sub><sup>+</sup>, 41% was stored as particulate nitrogen in the sediments



and, by difference, we estimated that the remaining 24% was denitrified. We also observed considerable uptake of nitrate by the sediments during winter months ( $1.1 \text{ g-at m}^{-2} \text{ y}^{-1}$ ), suggesting an additional source of annual denitrification, since this nitrate uptake was not accompanied by ammonium release back to the water column.

Possible power plant effects were summarized in Boynton et al. (1980b). In brief, some significant differences were observed in plankton metabolism and benthic nutrient fluxes which appeared to be lower and higher, respectively, at the power plant site. However, it was uncertain if observed differences were due to power plant operations or a normal feature of the estuary in this zone of strong longitudinal gradients. To resolve this point, metabolism-nutrient flux studies were continued through 1979-80 at the original stations and at stations upstream and downstream of the power plant.

#### Steam Electric Station Operations

The Chalk Point steam electric station (SES) is located in the estuarine portion of the Patuxent River about 37 km upstream of the river mouth. Chalk Point SES currently has three units. Units one and two, which began operation in 1964 and 1965, respectively, are rated at 355 MWe each. Unit three, which began operations in 1975, is rated at 660 MWe. A fourth unit, similar to unit three, is currently under construction. Two methods of condenser cooling are employed at Chalk Point. Intake water for Units one and two enters the plant from an embayment on the south side of the plant at Swanson Creek. A nine-foot deep intake channel leads from the river channel to the embayment. Each unit extracts a maximum of  $946 \text{ m}^3 \text{ min}^{-1}$  of river water from the embayment

for heat exchange. An additional  $946 \text{ m}^3 \text{ min}^{-1}$  are removed from the embayment during summer months and added directly to the head of the discharge canal to provide temperature reduction of discharge water in compliance with Maryland water quality regulations (Nihursky and Boynton, 1978). The discharge canal enters the Patuxent upstream of the plant at Eagle Harbor, and is about 1900 m in length. Unit three (as will unit four) employs a semi-closed hyperbolic cooling tower system for cooling condenser water. This cooling process removes up to  $20 \text{ m}^3 \text{ min}^{-1}$  of water from the discharge canal to replace evaporation losses within the tower. Actual withdrawal rates of cooling water from the embayment vary with water temperature, salinity, and relative humidity.

Operations of SES such as Chalk Point can influence community metabolism and nutrient cycling via several mechanisms. Entrainment of dissolved and particulate materials through cooling systems can produce increases in rates of mineralization due to mechanical, thermal and biocide effects. Similarly, damage to phytoplankton can result in decreased photosynthetic rates. An increase in the rate of deposition of material coming from discharge waters can locally increase rates of benthic metabolism and nutrient remineralization.

In this study these potential impacts were evaluated in the following component programs. In each program sampling stations were distributed so as to document processes at the plant site and at locations upstream and downstream to separate power plant effects from those associated with normal longitudinal estuarine gradients.

<u>Power Plant Process</u>	<u>Potential Impact</u>	<u>Evaluation Technique</u>
I. Entrainment/ Biocides	(a) Change in photosynthetic capacity (b) Change in respiration/ production characteristics (c) Shifts in nutrient speciation	Water Column photosynthesis Water column and benthic metabolism Water quality measurements
II. Thermal loading Biocides	(d) Change in metabolic rates (e) Change in sediment re- mineralization rates	as in (a) and (b) Benthic flux measurements
III. Deposition	(f) Change in benthic respiration (g) Change in benthic remineralization (h) Change in light characteristics (i) Changes in deposition/ resuspension rates	as in (b) as in (e) Light measurements Sediment traps

### Description of Study

This study had two major objectives. The first was to conduct in-situ studies of major carbon, nitrogen, phosphorus and sediment rate processes such as sedimentation, resuspension, benthic fluxes of nutrients and organic matter and water column metabolism. The second objective was to develop budgets of nitrogen, phosphorus and carbon in 3 distinctive areas of the estuary, including the area potentially influenced by power plant operations.

To meet these objectives, stations were located in the near-field at the terminus of the discharge canal (Potts Point) and in the far-field near Buena Vista, approximately 6 km downstream of Potts Point (Fig. 1). Because there are strong longitudinal gradients in water quality variables (see, for example, Mihursky and Boynton, 1978), and perhaps other factors as well in this zone of the estuary, two additional stations were selected, one above the

power plant at Jones Point and one downstream of the plant at Marsh Point. The two stations in the vicinity of the plant were in the high turbidity, low salinity zone while the upstream and downstream stations were in the tidal-fresh and moderately stratified mesohaline zones, respectively. Thus, the distribution of sampling stations reflected 3 distinctive regions of the estuary and included a site in the immediate vicinity of the power plant.

At these primary locations, estimates were made of water column metabolism, benthic respiration, benthic nutrient fluxes, water quality characteristics of the sediments and overlying waters, light extinction and sedimentation and resuspension rates of particulate materials (Table 1). On a quarterly basis, water quality surveys were conducted along the mainstem of the estuary with 10 stations arranged from the mouth to river kilometer 72. Four, 13-hour tidal surveys were also conducted at 3 different stations to estimate tidally-induced variability in water quality variables collected during synoptic longitudinal surveys. Finally, 4 water quality surveys were conducted in the intake-discharge canals at the power plant and were designed to allow estimates to be made of net gains and losses of carbon, nitrogen and phosphorus related to power plant operation. These measurements were made in 1980-1981 (Table 1).

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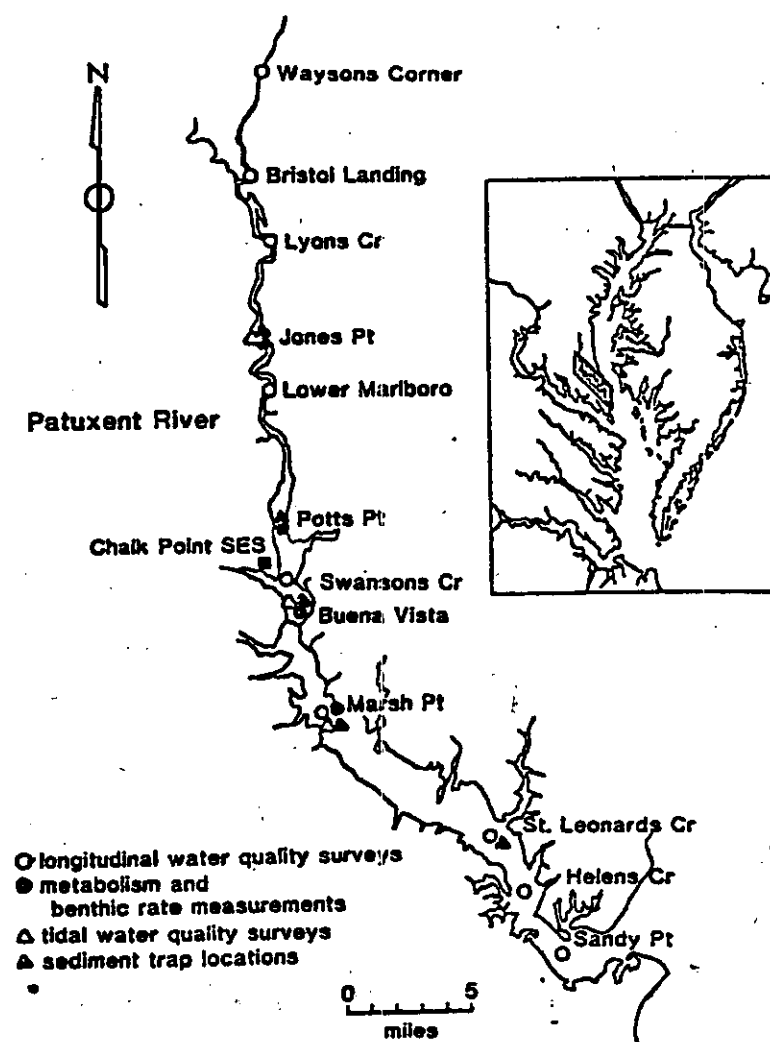


Figure 1. Location of sampling stations and description of measurements made in the Patuxent River estuary, July 1979 - May 1980.

Table 1. Measurement frequency of major study elements during 1979-1980

Study Element	1979						1980						
	J	A	S	O	N	D	J	F	M	A	M	J	J
* Benthic Fluxes	X		X	X		X		X			X		
* Plankton Metabolism	X		X	X		X		X			X		
* Longitudinal Surveys	X			X				X			X		
* Tidal Surveys		X			X					X		X	
* Sediment Trap Deployments		X	X	X	X			X	X	X	X	X	X
* Discharge Canal Surveys <sup>1</sup>					X				X		X	X	

<sup>1</sup> Conducted in 1980 and 1981

## METHODS

### Chemical Analyses

The chemical factors monitored during field studies and routine analytical methods (including percent recovery, day-to-day standard replication and instrumentation) are summarized in Tables 2 and 3, respectively. Dissolved nutrient fractions were determined from water samples filtered through 1.2  $\mu$ m filter pads (Whatman GF/C), frozen immediately and stored for analysis. Analyses took place within 4 weeks of the collection date.

Estimates of particulate nitrogen (PN) and carbon (PC) concentrations in the water column were obtained by filtering known quantities of water through precombusted (550°C for 1 hr) GF/C filter pads, which were then dessicated at 40°C for 48 hrs and frozen. Immediately prior to analysis, samples were thawed and dessicated for a second time for 24 hrs at 40°C and then analyzed using a Perkin-Elmer 240B elemental analyzer. Water column particulate phosphorus was determined as the difference between total phosphorus in an unfiltered water sample, and total dissolved phosphorus.

Sediment particulate carbon and nitrogen concentrations were estimated by combustion (at 900°C) of a known weight of dry sediment. Estimates of sediment particulate phosphorus concentrations were obtained by persulfate digestion of a known weight of dry sediment.

Chlorophyll a was determined by fluorometry, following extraction with a 90% acetone solution.

Table 2. Nutrient and particulate parameters monitored by the various field studies at Chalk Point S. E. S. July 1979 - July 1980.

Nutrients and Particulate	Benthic Nutrient flux	Longitudinal Profiles	Tidal Profiles	Interstitial water	Surface Sediment
$\text{NO}_2 - \text{N}$	X	X	X	X	
$\text{NO}_3 + \text{NO}_2 + \text{N}$	X	X	X	X	
$\text{PO}_4^{+}$	X	X	X	X	
$\text{DON}^{1/}$	X	X	X	X	
$\text{DOP}^{2/}$	X	X	X	X	
PC		X	X	X	X
PN		X	X	X	X
$\text{PP}^{3/}$		X	X	X	X
TP		X	X	X	X
Active/Total chlorophyll <u>a</u>		X	X		X

1/ Dissolved organic nitrogen determined as the difference of TDN minus the summed inorganic N.

2/ Dissolved organic phosphorous determined as the difference of TDP minus  $\text{PO}_4^{+}$

3/ Particulate phosphorus determined as the difference of total phosphorus in an unfiltered sample minus TDP after filtration.



Table 3. Summary of nutrient analysis methods, giving percent recovery, standard replication and instrumentation used.

Nutrient	Recovery, %	Analysis Specification		
		Std. Replication Day to Day %	Instrumentation	Reference <sup>4/</sup>
NO <sub>2</sub> -N	100	+5	Auto Analyster II	Technicon Industrial Systems (1972)
NO <sub>2</sub> +NO <sub>3</sub> as N	94	"	" "	" "
NH <sub>3</sub> -N	92	"	" "	Technicon Industrial Systems (1978)
DON <sup>1/</sup>	100	"	" "	D'Elia et al. (1977)
DIP	98	"	" "	Technicon Industrial Systems (1973)
DOP <sup>2/</sup>	96	"	" "	Glibert, P. M. et al. (unpublished)
TP	96	"	" "	" "
DOC			Infrared Analyzer	Menzel and Vaccaro (1964)
Active/total	90	+20 <sup>3/</sup>	Fluorometer	Holm-Hansen, et al. (1965)
PC			Elemental Analyzer	Hobson, L. A. and D. W. Menzel (1969)
PN			"	" "
TP (sediment)	97-102	5	Spectrophotometer	Anderson, J. M. (1976)

<sup>1/</sup> Dissolved organic nitrogen determined as total dissolved nitrogen (TDN) after filtration and reported as the difference of summed inorganic N and TDN.

<sup>2/</sup> Dissolved organic phosphorus determined as TDP after filtration and reported as the difference of DIP and TDP.

<sup>3/</sup> For chlorophyll a only.

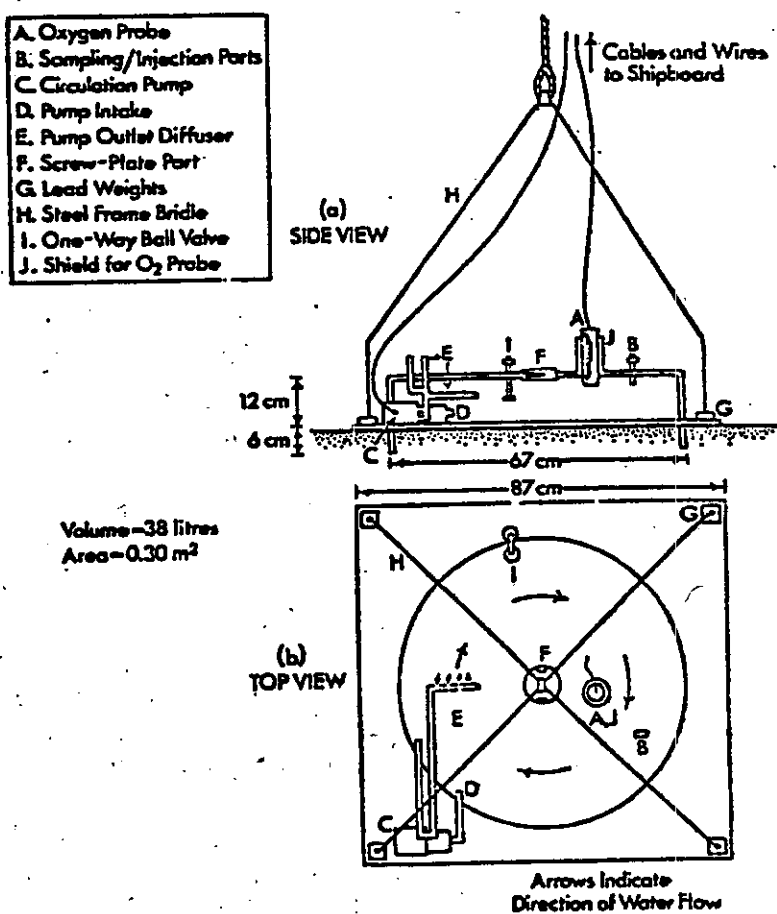
<sup>4/</sup> Complete citations given in Osborne et al. (1979).

Suspended load (seston) was estimated by filtering a known volume of raw water through pre-weighed GF/C filter pads, drying to a constant weight, and weighing. Concentrations were then determined by difference and adjusted for the volume filtered.

#### Sediment Oxygen Demand

In-situ measurements of sediment oxygen demand were made using 3-4 opaque Plexiglas chambers (6 mm gauge), cylindrical in shape with a circular area of  $0.3 \text{ m}^2$ , a height of 18 cm (12 cm above the sediment surface) and an enclosed volume of 38 L. The basic design is depicted in Figure 2, illustrating several pertinent details. Operational characteristics of the chambers has been reported by Boynton et al. (1981). The chambers were mounted on a square flange (87 cm wide) which provided a base to rest on the sediment surface, and which allowed 6 cm of the chamber's edge to penetrate the sediments, insuring proper seal. A dissolved oxygen (DO) and temperature probe (YSI 5739) was mounted in the top of each chamber. A submersible pump (Teal IP681) regulated by a rheostat (Powerstat 3PN116B) with a diffuser outlet provided a reasonably uniform clockwise circulation under the chambers. Probes and pumps were connected to shipboard via cables. The pumping/circulation system was calibrated in the laboratory with the chambers inverted. Dye injection experiments allowed a visual check on the uniformity of flow, while direct current measurements using an electromagnetic induction meter (Marsh-McBirney Model 201) enabled us to calculate mean velocity at a given rheostat setting. The circulation pattern was characterized by a near-laminar flow regime except in the mixing zone 5-8 cm out from the diffuser outlet and at the chamber's

Fig. 2. Schematic diagram of experimental chamber design used for in-situ measurements of fluxes across the sediment/water interface



center (10-12 cm diameter) where turbulent conditions prevailed. This turbulent region was somewhat larger at higher flows, but never exceeded about 8% of the enclosed area. During calibration we made 40-60 discrete velocity measurements in the three-dimensional chamber volume at several rheostat settings. Actual calibrated mean velocities were slightly lower than the nominal rates but the two values never varied by more than 10%, when, for instance, at a nominal  $10 \text{ cm s}^{-1}$  setting, velocities ranged from about  $7.1\text{-}11.9 \text{ cm s}^{-1}$ , with a coefficient of variation of about 15% around a mean velocity of  $9.2 \text{ cm s}^{-1}$ . All field measurements were made using a nominal velocity of  $10 \text{ cm s}^{-1}$ . Since calibration work was done with the chambers inverted rather than on the sediment surface, we expect that current velocities inside the chambers during field measurements were somewhat slower than measured velocities and the circulation pattern somewhat more turbulent due to the frictional drag of the sediments.

During each field survey, simultaneous measurements were obtained using 3-4 chambers placed on the bottom within about 10 m of one another. Chambers were placed in the water, voided of gases, slowly lowered to the bottom, and gently pushed (by SCUBA divers) into the sediments to the height of the flanges. After an equilibration period (15 min), the pumps were set at the nominal velocity of  $10 \text{ cm s}^{-1}$  and turned on. Temperature and DO in chambers were recorded at 10-min intervals for a period of about 1-3 h, during which time changes in DO ranged from about 0.5-2.0 ppm and minimum concentration never fell below 3.0 ppm. Oxygen consumption in triplicate opaque BOD bottles served as a control to account for plankton respiration. Sediment oxygen demand was calculated using the slope of a regression line fitted through

oxygen concentration data points plotted against time. Adjustments were made for the volume and area of the chambers and results expressed as  $\text{g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ .

Nutrient fluxes across the sediment-water interface were also measured at each station during each cruise. Concentrations of the following nutrient species were measured for water samples from the chambers, taken throughout incubation periods (1-3h): nitrate ( $\text{NO}_3^-$ ); nitrite ( $\text{NO}_2^-$ ); ammonium ( $\text{NH}_4^+$ ); dissolved inorganic phosphorus (DIP); dissolved organic phosphorus (DOP); total phosphorus (TP); and dissolved organic nitrogen (DON). Measurements of particulate carbon, particulate nitrogen, chlorophyll *a* and seston were also made for the water column at each station. Nutrient flux across the sediment-water interface was estimated by following the concentration changes within a chamber throughout a measurement using the mean rate as calculated from a fitted regression time. Nutrient concentration changes generally appeared linear over sampling periods. Nutrient changes in triplicate dark BOD bottles incubated at ambient temperature were used as an estimate of water column nutrient changes.

#### Water Column Metabolism

Estimates of water column photosynthesis and respiration (metabolism) were made using the oxygen light and dark bottle method. Depending on the turbidity conditions encountered on the day of the measurement; triplicate 300-ml light bottles filled with surface water (0.5-1 m depth) were suspended at depths approximating 90%, 60% and 10% of surface insolation. A single group of three, 300-ml dark bottles was suspended at the 10 percent level. Bottles were deployed at dawn and retrieved at dusk. Triplicate samples were

also taken to establish the initial dissolved oxygen concentration at the start of the incubation period. Gross primary production (P) (volumetric) was calculated as the difference between oxygen concentrations in the light and dark bottles after incubation. Total diel respiration (R) (volumetric) and night respiration (R<sub>N</sub>) were calculated as the difference between initial and dark bottles. Apparent primary production (P<sub>A</sub>) (volumetric) was then calculated as the difference between the initial and light bottles. Areal estimates of water column photosynthesis and respiration were calculated by plotting volumetric rates at the depths of incubation and graphically integrating the area enclosed by the curve. Volumetric rates for respiration were extended to the bottom at each station while volumetric rates for photosynthesis were extended to the bottom or to the depth where surface insolation was reduced to 1 percent.

The following relationships were used to estimate P, P<sub>A</sub>, R, R<sub>N</sub>:

$$P = \frac{L - D}{(\text{hrs of Incubation})} \times \text{daylight hrs}; \quad P_A = \frac{L - I}{(\text{hrs of Incubation})} \times \text{daylight hrs}$$

$$R = \frac{I - D}{(\text{hrs of Incubation})} \times 24; \quad R_N = \frac{-D}{(\text{hrs of Incubation})} \times \text{Hrs of darkness}$$

Extinction coefficients were estimated using a quantum meter (LI-COR 185A) utilizing a LI-192S submersible quantum sensor. The measured 1% light level was used to delineate the depth of active photosynthesis.

#### Gas Transfer

On several occasions estimates of gas transfer across the air-water interface were made using the floating dome method of Copeland and Duffer

(1964) as modified by Hall (1970). Measurements were taken in water depths of 3 and 6 m having current velocities between 0 and 37 cm s<sup>-1</sup>. This method involves filling a rigid floating dome (volume = 17.7 L; area = 0.20 m<sup>2</sup>) with nitrogen gas at atmospheric pressure and recording the increase in oxygen over time with an oxygen probe placed inside the dome. The observed rate was corrected for ambient temperature and pressure conditions and expressed as a coefficient in units of g O<sub>2</sub> m<sup>-2</sup> hr<sup>-1</sup> at 100% saturation deficit.

#### Water Column Characterization

On a quarterly basis, coinciding with sediment oxygen demand measurements, water column water quality variables (Table 2) were estimated using two different sampling strategies: longitudinal profiles, and tidal profiles.

Longitudinal profiles involved measuring dissolved and particulate nutrient concentrations along the entire length of the study area on a given day. Upon arriving on station, temperature-salinity measurements were obtained at 1-3 m intervals. In there was evidence of stratification, multiple samples, representative of the various unmixed layers were obtained. If no stratification was evident, a single water sample from mid-depth was obtained. Samples were filtered, frozen and transported to the lab for analysis.

Tidal profiles involved measuring dissolved and particulate nutrient concentrations in the water column over a tidal cycle. A single station was occupied for a 13-hr period. At hourly intervals temperature-salinity

profiles were obtained at 1-meter intervals. If stratification was evident, multiple water samples were obtained. A lack of stratification resulted in a mid-depth sample only. Water samples were filtered, frozen and transported to the lab for analysis. In addition, wind speed, direction, current velocity and direction and secchi disc measurements were obtained.

#### Sediment Nutrients

On a quarterly basis, dissolved and particulate fractions of carbon, nitrogen and phosphorus were monitored at each benthic respiration station and coincided with benthic flux surveys done in July, October, February and May. Interstitial water concentrations and particulate fractions were profiled using a single diver-operated 7.1-cm diameter x 50-cm plastic core at each station.

Undisturbed cores were transported to the laboratory, placed in a nitrogen-filled glove box, extruded by water pressure acting on a piston, and sectioned at 2-cm intervals from 0-10 cm and at 5-cm intervals beyond the 10-cm level. Interstitial water was extracted by centrifugation at 40°C and 1000 gravities for 30 min (Engler et al., 1977). The resulting water was then filtered through a 1.2- $\mu$ m filter and aliquots frozen until analysis.

Surface samples of particulate carbon and nitrogen were obtained using a diver-operated syringe corer (sampling area of 1.13 cm<sup>2</sup> inserted to a depth of 1 cm). Sub-samples were taken for carbon, nitrogen and phosphorus. These were frozen, dessicated and later analyzed. Fractions for chlorophyll a were placed in a 90% acetone solution, extracted and analytically treated as a



water column chlorophyll a sample.

#### Sedimentation and Resuspension

The rates of sedimentation and resuspension were estimated using cylindrical PVC cups with an effective collecting area of 45.6 cm<sup>2</sup> and a height-to-width ratio of 2.7. Cups were firmly attached to a taut moored buoy and left unattended for the sampling period. Blanks, attached at specific intervals, consisted of cups left open at either end until retrieval of the string. Figure 3 details a typical sediment trap string.

All of the traps were sampled as follows: a SCUBA-equipped diver would carefully descend the trap, capping each cup with a polyethelene cap. Upon reaching the bottom a tag line was placed upon the anchor and the entire string released. After recovery, each cup was removed; the string was cleared of fouling, clean cups placed on the line and reattached to its anchor. Recovered sediment trap cups were then placed in chilled estuarine water and transported to the lab for processing and analysis.

In the laboratory each cup was partially decanted into a 2 L beaker. The cup was then scrubbed vigorously on the inside, care being taken to prevent any material from the lip entering the sample. This portion was then added to the beaker. A final rinse of the cup and brush with distilled water facilitated the removal of the last of the sediment. The total volume of the sample was then determined. The sample was then suspended using a magnetic stirrer. Aliquots were taken and filtered through pre-weighed, pre-combusted GF/C filter pads. Triplicate samples were taken from each cup for the

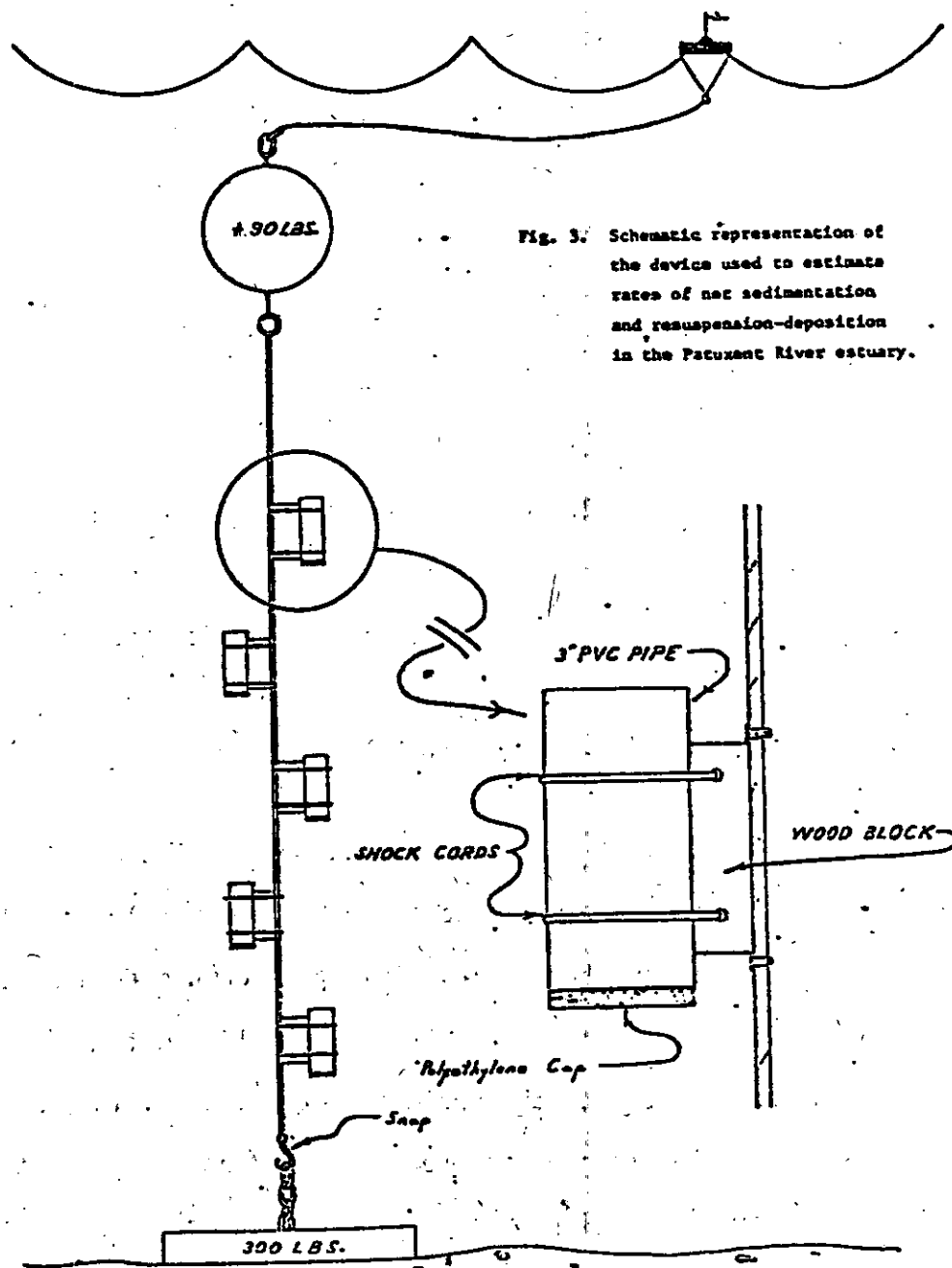
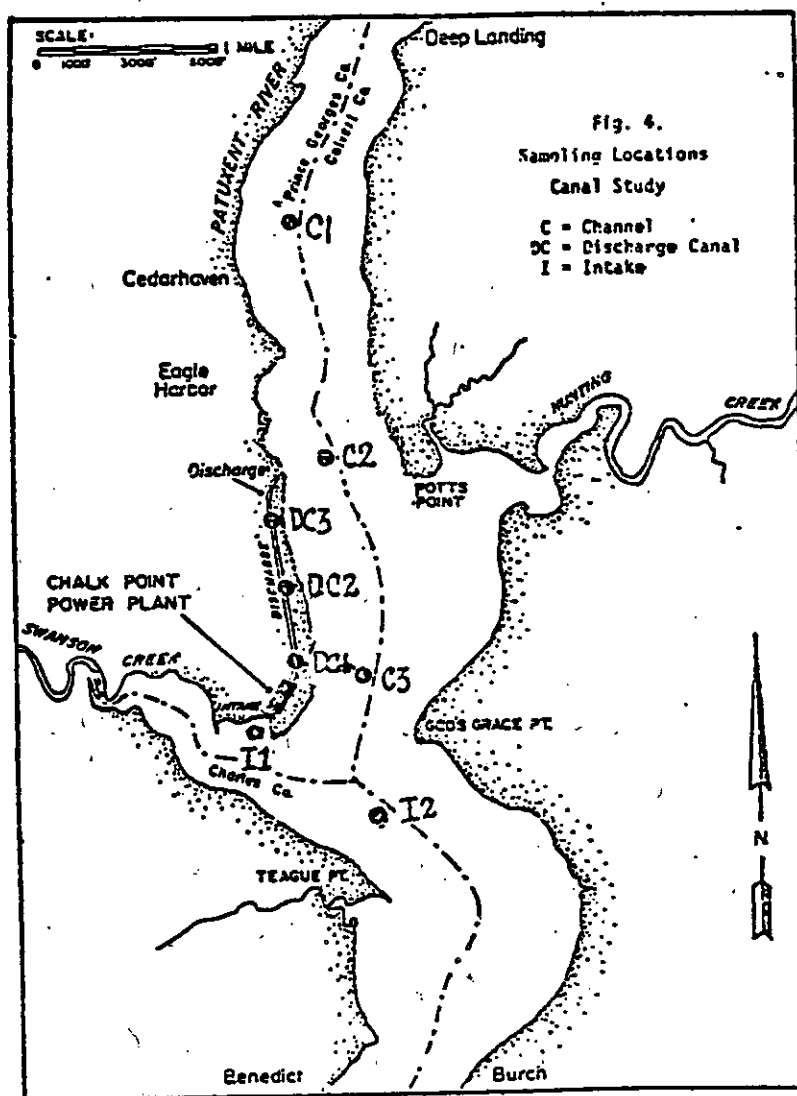


Fig. 3. Schematic representation of the device used to estimate rates of net sedimentation and resuspension-deposition in the Patuxent River estuary.

determination of total sediment, particulate carbon and particulate nitrogen. A second set of duplicate pads was also taken for the determination of total phosphorus. The data were converted to units of mass/m<sup>2</sup>/day. Bottom cups were used to calculate resuspension rates while mean values of several near-surface cups were generally used to calculate net sedimentation. Surface values were extrapolated to the bottom assuming a linear increase in material with increases in depth.

### Canal Studies

On four occasions (Nov. 13, 1980; March 24, May 5, and June 6, 1981) water quality samples were taken at locations indicated in Fig. 4 to obtain estimates of the magnitude of water quality changes due to passage through the power plant and transit down the discharge canal. Additional stations were also sampled so that the mass of material exiting the discharge canal could be compared to that being transported by the river. To make these estimates, water column integrated samples were taken at stations shown in Fig. 4 and analyzed for the normal spectrum of constituents using analytical methods described earlier. Raw data are given in Appendix 1. Sampling was initiated shortly after the tide had started ebbing during each survey. On most surveys stations I1 and I2 were sampled first and then a dye spot was introduced at DC1 and a water sample was taken at that station. The dye patch was then followed down the canal and vertically integrated water samples taken when the patch reached stations DC2 and DC3. Finally, samples were collected at stations C1, C2, and C3.



## NITROGEN AND PHOSPHORUS DYNAMICS

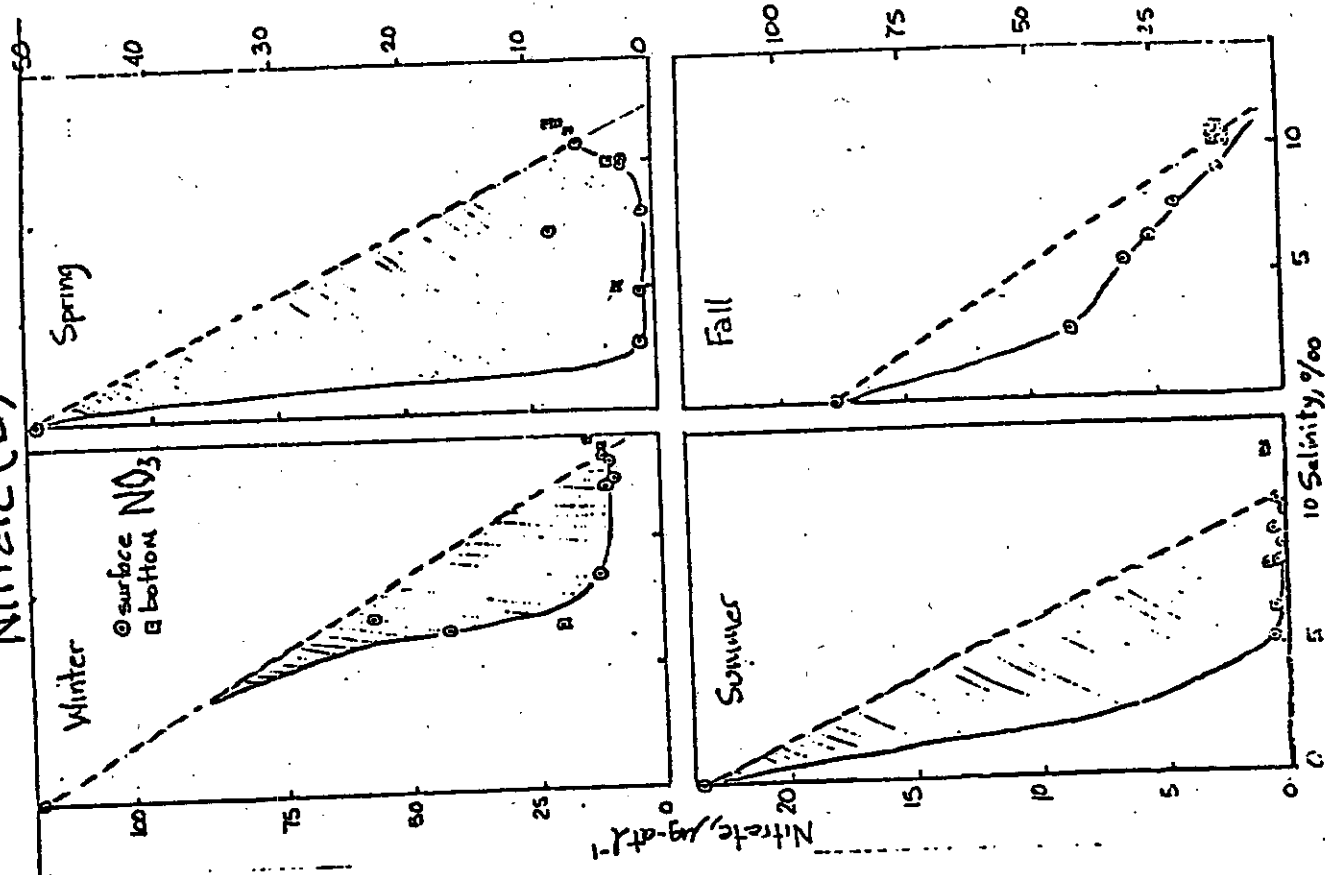
In this section we consider the spatial and temporal characteristics of nitrogen and phosphorus compounds contained in the water column and sediments. In addition, net fluxes of nutrients between sediments and overlying waters are evaluated.

### Mixing Diagrams

The broad pattern of interaction between estuarine water quality concentrations and physical characteristics such as flushing can be inferred from geographical presentations of water column concentrations plotted against associated salinities. With nutrient concentration on the vertical axis and salinity on the horizontal axis, a straight-line relationship suggests that there is no net loss or gain of the material as it passes from the head to the mouth of the estuary, assuming that the nutrient source is at the head of the estuary. A concave line indicates that the estuary is serving as a nutrient sink, while a convex line indicates a net source of nutrients. We have developed seasonal plots for major nutrient species ( $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , Total N, ortho- $\text{PO}_4$ , Total P) and these are presented in Figures 5 and 6.

The seasonal patterns of ammonium concentration vs. salinity appeared complex, showing both source (spring) and sink (summer and winter) characteristics (Fig. 5). In addition, concentrations at low salinity stations decreased by about an order of magnitude between winter and spring and remained low through the remainder of the year. High ammonium concentrations ( $32 \mu\text{g-at l}^{-1}$ ) coincided with periods of peak river flow,

# Nitrate (b)



# Ammonium (a)

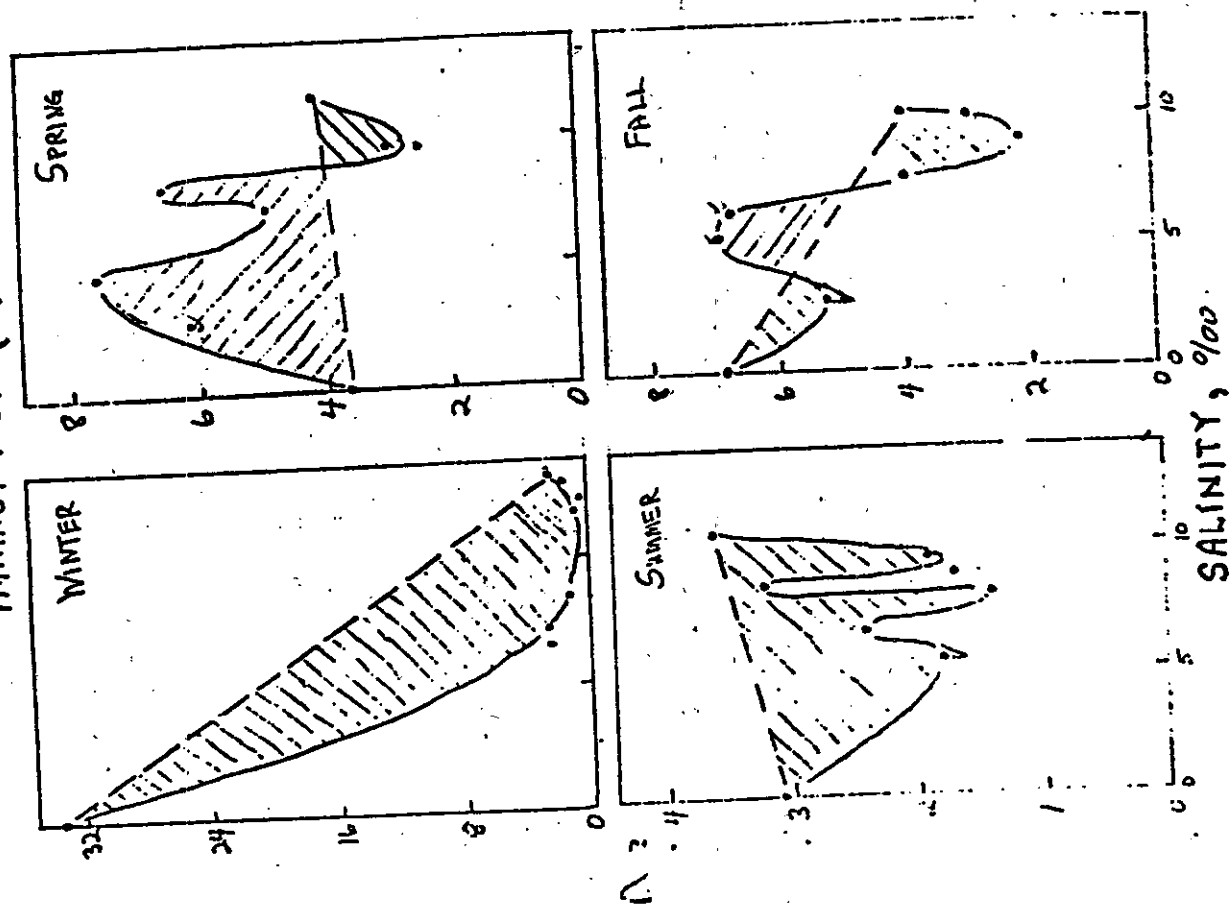


Fig. 5. Seasonal concentration-salinity plots of selected water quality variables collected in the Patuxent Estuary in 1979-80. (a) ammonium, (b) nitrate, (c) total nitrogen.

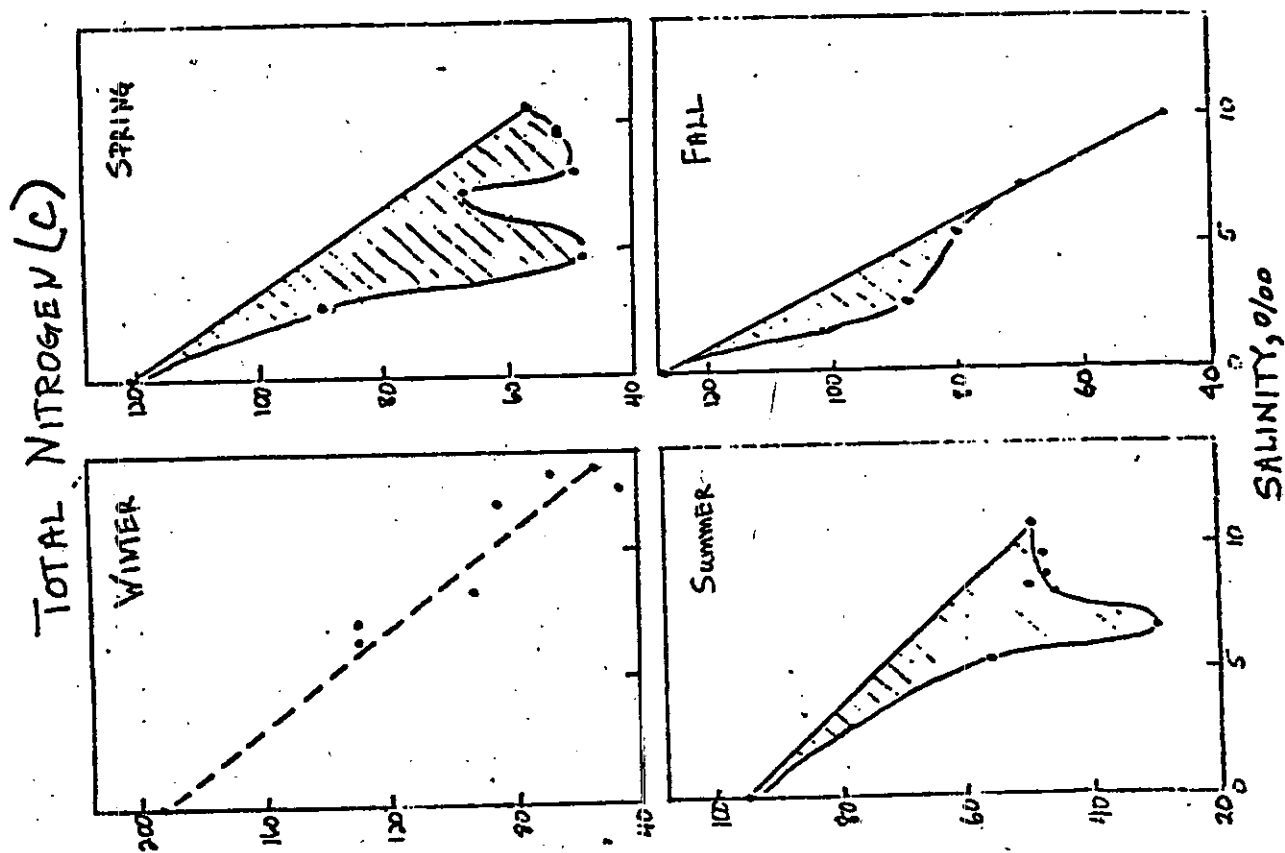


Fig. 5 (continued)

suggesting that the source of this ammonium was from non-point drainage from the water shed. The relatively sharp decline in ammonium concentrations during the winter may have resulted from nitrification processes rather than uptake by phytoplankton because phytoplankton production was low at that time of year. Unfortunately, we have no direct measurements of nitrification rates in this zone of the river during winter months to confirm this speculation. During the remainder of the year ammonium concentrations were lower and varied in a non-conservative fashion most possibly reflecting substantial uptake by phytoplankton (sink) and recycling in both the water column and sediments (source). In all seasons of the year, there was a tendency for concentrations to increase near the confluence of the estuary with Chesapeake Bay, probably reflecting the input of deeper, ammonium-rich water from the Bay to the estuary.

During all seasons of the year (but most evident in the spring and summer) nitrate displayed sink characteristics, particularly in the low salinity (0-5 ppt) zone of the estuary (Fig. 5). It is unfortunate that more stations were not sampled in this zone of strong gradients for purposes of more clearly defining possible mechanisms. In any case, at least two mechanisms may be involved including phytoplankton utilization of nitrate and denitrification. Several lines of evidence suggest that denitrification was the primary factor causing this rapid decrease in concentration although phytoplankton uptake may well be a contributing factor. Recent measurements of denitrification by Jenkins (pers. comm.) in the area of Benedict indicated that rates are high, at least in the spring. Furthermore, in-situ benthic nutrient flux measurements consistently show nitrate entering the sediments at



high rates (  $100-500 \mu\text{g-at m}^{-2} \text{ hr}^{-1}$ ) when nitrate is available in the overlying water. During the winter-spring, nitrate fluxes into sediments are not accompanied by comparable ammonium fluxes from sediments (indicative of nitrate reduction) or accumulation of nitrate in interstitial waters (nitrate storage). We have suggested elsewhere that denitrification may be a major term in nitrogen dynamics in at least some zones of this estuary (Boynton et al. 1980a).

Concentrations of total nitrogen ( $\text{NH}_4 + \text{NO}_2 + \text{NO}_3 + \text{DON} + \text{PN}$ ) displayed strong sink characteristics during the spring and summer; strong net changes were not so evident in the winter and fall (Fig. 5). In overview, it appeared that total nitrogen was conservatively diluted during the winter and fall and that a major fraction of this nitrogen was in dissolved and particulate organic forms. However, in the spring and summer, substantial nitrogen losses occurred beyond that which could be accounted for by changes in nitrate and ammonium. This decline appeared to be related to declines in both particulate and dissolved organic forms. Results from sediment trap studies (see following sections) support this observation in that sedimentation rates of particulate nitrogen were generally higher in the warm seasons of the year.

Dissolved inorganic phosphate exhibited depositional characteristics only in winter (Fig. 6). This is in sharp contrast to the data of Flemer et al. (1970) which indicated strong depositional features in all seasons. Data collected in 1980-81 exhibited a pattern similar to that which we observed (Sage, pers. comm.). In the present study, phosphate concentrations were at least 3 times lower than those observed by Flemer<sup>mc</sup> et al. (1970) and comparable

to those found in 1980-81. The lower concentrations we observed may have resulted from the low runoff and river discharge conditions present in 1979-80. In turn, deposition of dissolved phosphate may be related to adsorption processes in the estuary as suggested by Stirling and Normald (1977). Thus, when phosphate concentrations are high, deposition characteristics emerge; when upstream concentrations are low, only small changes occur along estuarine gradients.

Patterns of total phosphorus concentration were quite similar to those observed for dissolved inorganic phosphorus in the winter, spring and fall, although upstream concentrations were 2.5-6 times higher, indicating that particulate and dissolved organic phosphorus forms dominated the pattern in the water column (Fig. 6). However, in the summer there was a substantial peak in total phosphorus in the 5-8 ppt salinity region which occurred somewhat downstream of the summer inorganic phosphorus peak and which was mainly composed of particulate and dissolved organic forms. Within the context of concentration-salinity plots, there were no major deviations in concentration in the vicinity of the power plant discharge canal.

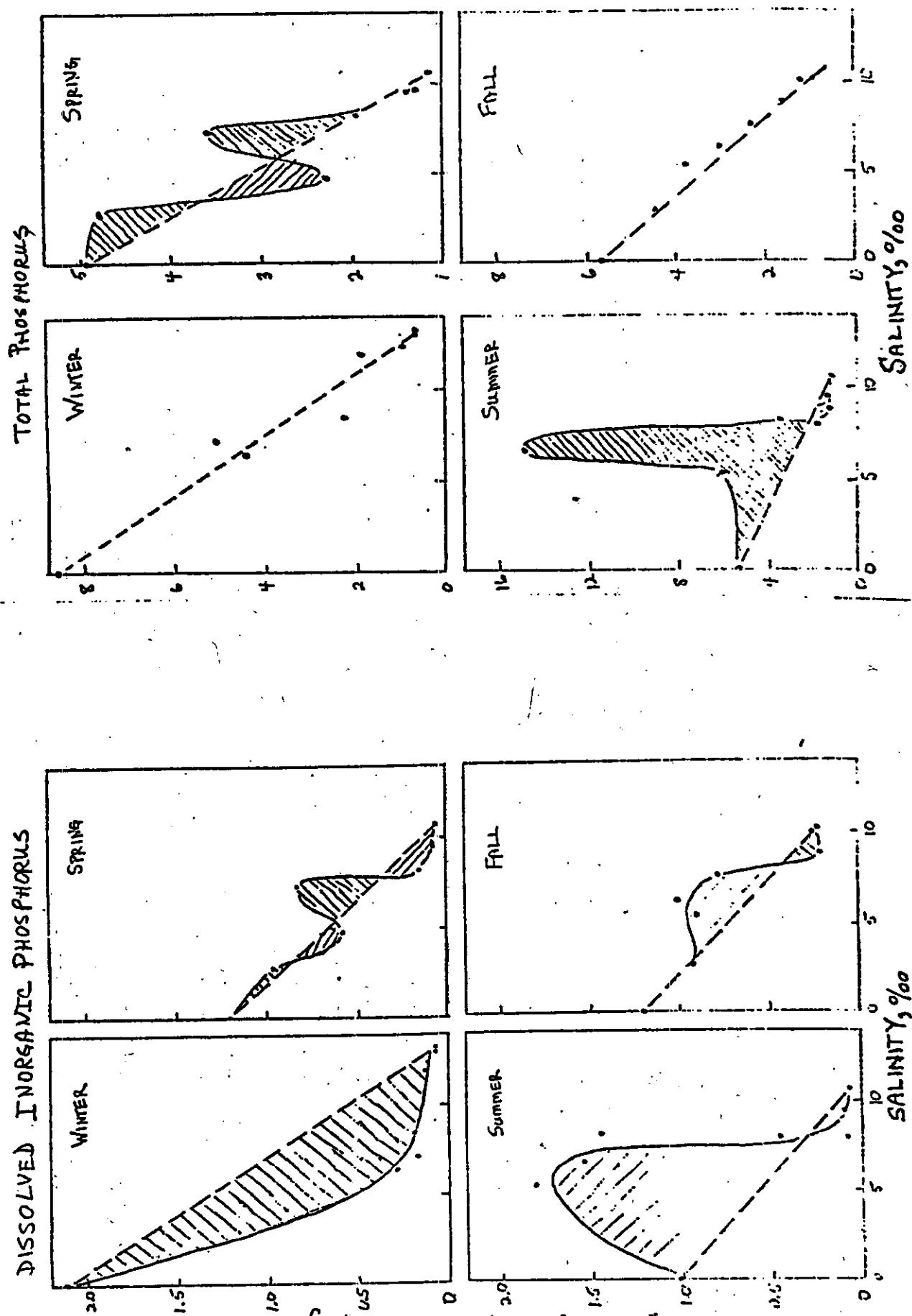


Fig. 6. Seasonal concentration-salinity plots of selected water quality variables collected in the Patuxent Estuary, 1979-80. (a) dissolved inorganic phosphorus (b) total phosphorus.

### Ratios of Nitrogen to Phosphorus

From both scientific and management standpoints, an important issue is whether nitrogen or phosphorus is the nutrient governing phytoplankton photosynthesis. A simple approach to this complex problem involves constructing ratios of nitrogen (usually the forms readily available to phytoplankton including  $\text{NO}_3^-$ ,  $\text{NO}_2^-$  and  $\text{NH}_4^+$ ) to phosphorus (usually dissolved inorganic phosphorus). Using the concept that most phytoplankton utilize N and P in ratio of about 16:1, examination of N:P ratios indicate which element would become limiting first. Thus, ratios below 16:1 suggest a potential nitrogen limitation while those above 16:1 suggest a phosphorus limitation. We have constructed time-space diagrams of N:P ratios for years in which data were available (Fig. 7).

Examination of the data presented in Fig. 7 indicates that there is considerable variability in N:P ratio distributions between different years. For example, in 1979-80, the region of low N:P ratios was restricted to the early fall period and extended over only a small section of the estuary. In strong contrast to this, low N:P ratios extended over broad regions of the estuary from early spring to late fall in 1969-70, while data from 1980-81 exhibited an intermediate pattern. The exact causes of year-to-year changes are unclear but, in a qualitative fashion, the spatial and temporal extent of low N:P ratios appears to be proportional to river flow. It is interesting to note that nitrogen is considerably more mobile than phosphorus in being transported via run-off to the river. We might expect, then, that in years of high run-off (and high river flow), N:P ratios would be higher than in lower

discharge years. In fact, the opposite appears to be the case. Since both the absolute concentrations of N and P and the relative availability of each are central to understanding phytoplankton dynamics and managing water quality in this estuary, the above issue deserves further investigation.

In all years for which data were available, the region around the Chalk Point power plant was in the zone of low N:P ratios. In 1979-80 and in 1980-81, zones of low N:P ratios extended from the power plant area downstream 15-20 km. However, in 1969-70, low N:P ratio zones extended both upstream and downstream of the power plant area. In a later section we explore possible roles of the power plant in modifying the relative abundance of nitrogen and phosphorus in the water column.

#### Variability in Water Column Nutrient Concentrations

In attempting to examine and characterize nutrient processes in an estuarine environment it is often necessary to deal with average or mean conditions. For example, it is a common practice to conduct "slack-water" surveys of water quality variables for use in modeling exercises. In this study we conducted several longitudinal surveys of water quality conditions to characterize zones and seasonal periods of regeneration, deposition, and transport to assist in confirming more detailed studies of such processes in only a few areas of the estuary. An obvious question concerns the accuracy of

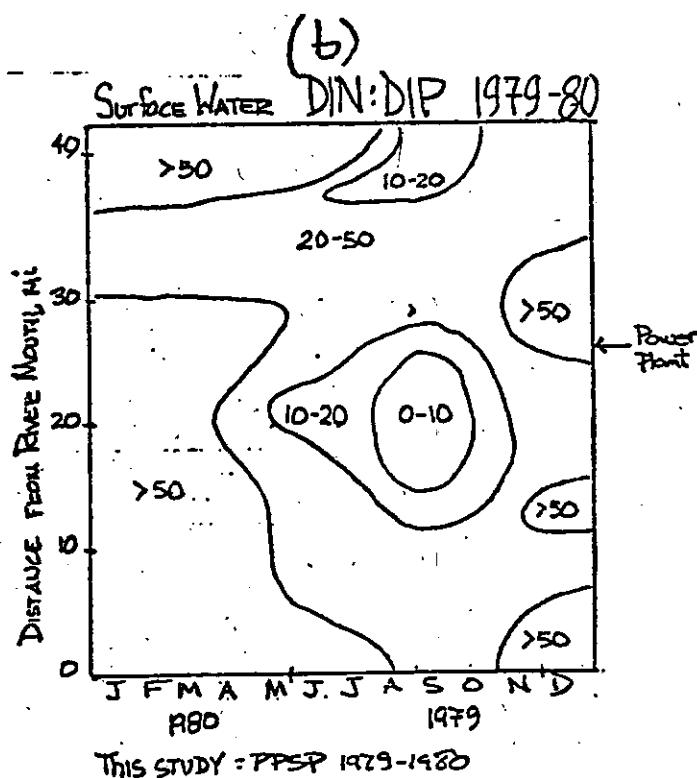
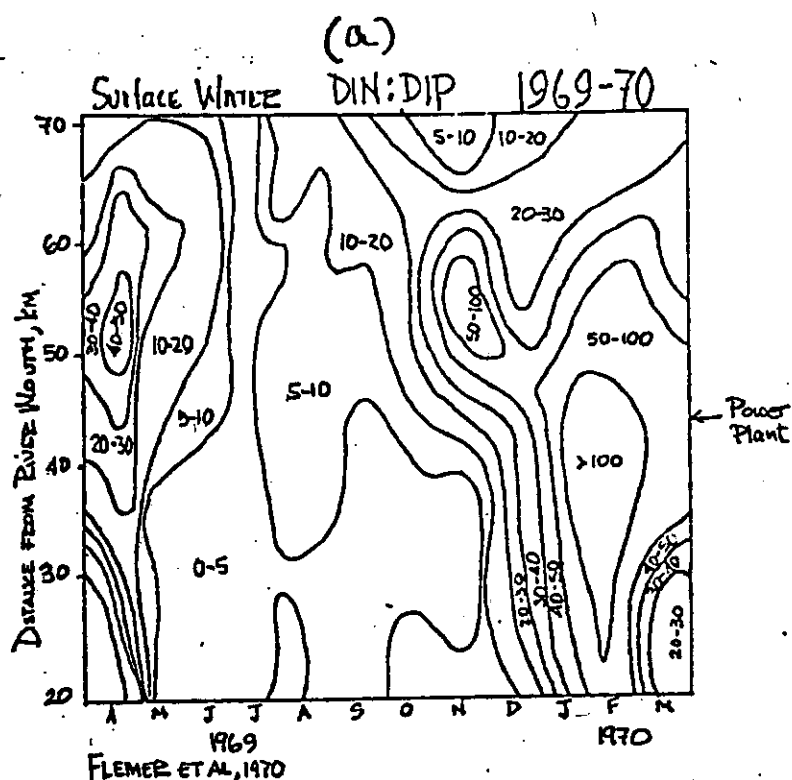


Fig. 7. Time-space diagrams displaying contours of nitrogen to phosphorus ratios for several different years. Nitrogen concentration included  $\text{NO}_3^-$ ,  $\text{NO}_2^-$  and  $\text{NH}_4^+$  while phosphorus included only DIP. (a) 1969-1970 period (surface water), (b) 1979-1980 period (surface water), (c) 1980-1981 period (surface water), (d) 1980-1981 period (bottom water).

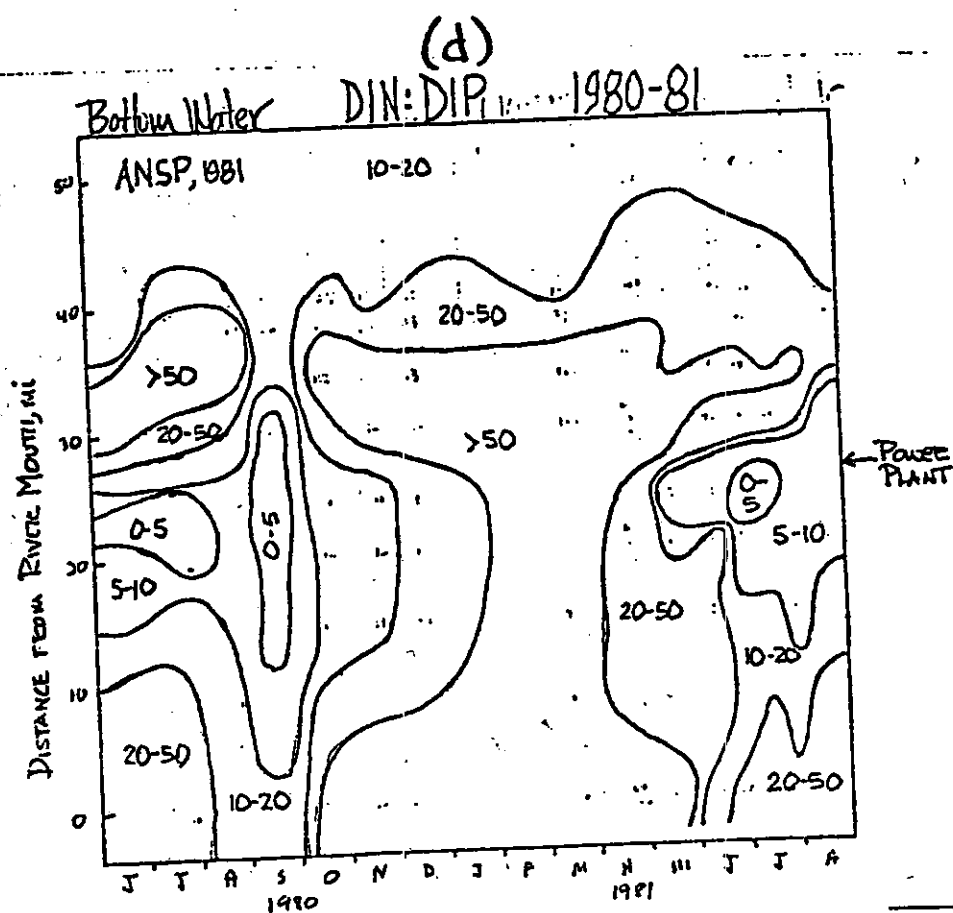
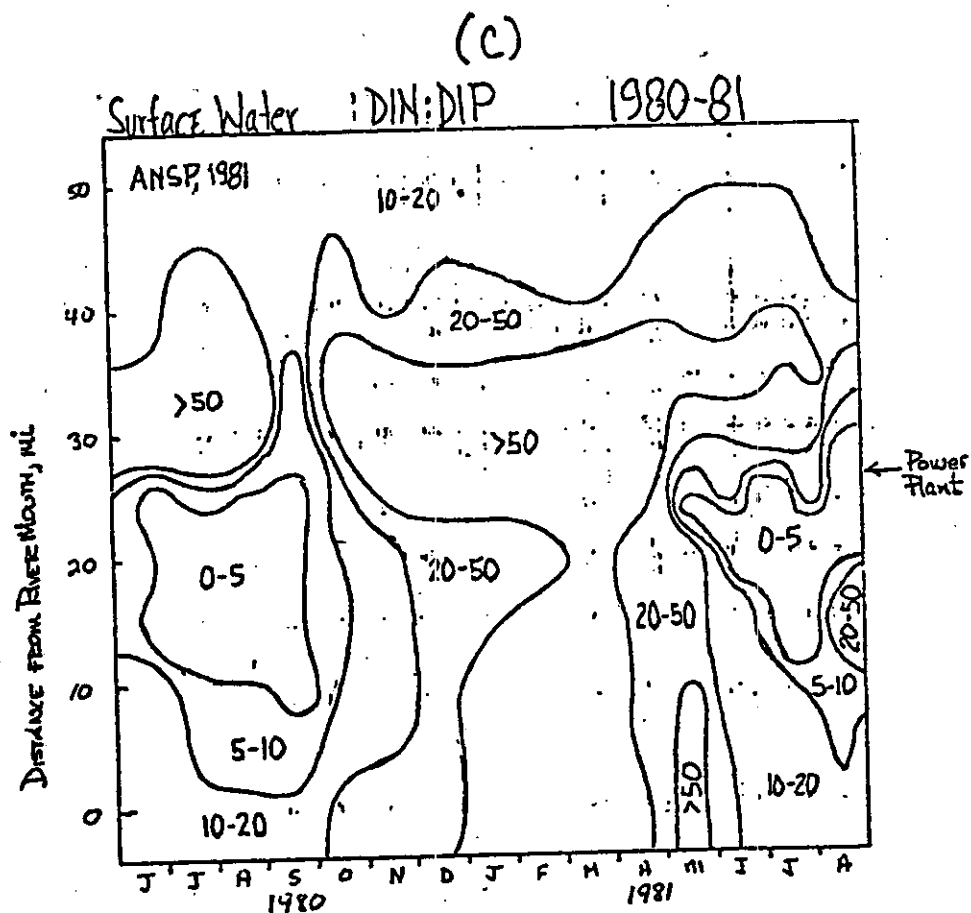


Fig. 7 (continued).

nutrient concentration measurements made only once at a given location in an estuary such as the Patuxent. To address this question, we conducted four 13-hr surveys at 3 locations in the Patuxent during 1979-80. Data from these studies were reported in Boynton et al. (1980b). We have since reviewed nutrient data relative to time-dependent concentration changes at these study sites and results are summarized in Table 4.

Visual inspection of data did not reveal strong or consistent changes in nutrient concentrations as a function of tidal stage. Rather, changes in concentration appeared to be erratic with respect to both magnitude and duration. To obtain an estimate of the variability of selected water quality variables over tidal cycles, coefficients of variation (C.V.) were calculated for each cruise (Table 4). In general, C.V.'s were in the range of 20-40% with occasional values above and below this range.

If additional analysis of this data set does not reveal strong tidal components related to nutrient concentration changes, then we would tend to view nutrient concentration distributions within tidal excursion distances as normally heterogeneous, perhaps reflecting strong variations in biological rate processes (e.g. phytoplankton assimilation, benthic nutrient recycling) controlling water quality variables.

While it appears that single estimates of nutrient concentrations exhibit considerable variability over short time periods (6-12 hrs), this does not substantially restrict the use of such data for some purposes. For example, concentrations of many nutrients change by as much as an order of magnitude over distances of only several miles (Figs. 5 and 6). Changes in



Table 4. Variability in selected water quality variables as a function of time (over one tidal cycle) expressed in terms of coefficient of variation

<u>Variable</u>	<u>LOCATION AND DATE</u>			
	<u>Buena Vista</u> <u>8/2/79</u>	<u>Jones Pt.</u> <u>11/12/79</u>	<u>Marsh Pt.</u> <u>4/11/80</u>	<u>Jones Pt.</u> <u>6/28/80</u>
Physical Factors	---			
Temperature	---	33.9	14.4	0
Salinity	---	0	19.3	0
Seston	24.7	58.2	21.8	21.1
Chemical Factors				
Dissolved Oxygen	---	28.7	8.0	7.4
Nitrate	126.7	8.0	17.3	11.9
Nitrite	26.9	10.3	15.0	16.1
Ammonium	27.0	16.1	19.2	31.1
Phosphate	8.7	10.4	23.3	33.6
Biological Factors				
POC	24.3	---	29.3	31.3
PON	46.4	---	32.4	36.4
POP	22.5	27.2	85.0	27.2
Chlorophyll	---	17.7	82.1	---

concentration over tidal periods is thus small compared to those observed over small spatial distances and thus tidal variability does not appreciably change conclusions reached by examination of salinity-nutrient concentration plots. Finally, changes in nutrient concentrations over longer time periods (days-weeks) is much greater than those observed over tidal periods and far greater between seasons at some locations in the estuary (Appendix 2).

#### Particulate Nitrogen and Phosphorus in Sediments

Sediment cores were obtained at each primary sampling station on 3 occasions in 1979-80 to (1) determine the amount of particulate nitrogen (PN) and particulate phosphorus (PP) stored in the sediments and (2) to explore possible relationships between PN and PP storage and observed nutrient fluxes. Data are displayed in Fig. 8 and in Appendix 3.

Several interesting features are evident in this presentation. First, the average amount of PN (expressed as  $\text{g m}^{-2} \text{ cm}^{-1}$ ) steadily increased from downriver to upriver stations (MP=9  $\text{g m}^{-2} \text{ cm}^{-1}$ ; BV=10  $\text{g m}^{-2} \text{ cm}^{-1}$ ; PP=16  $\text{g m}^{-2} \text{ cm}^{-1}$ ; JP=20  $\text{g m}^{-2} \text{ cm}^{-1}$ ). Concentrations of PP displayed an even stronger trend toward higher concentrations at upriver stations (MP=2.5  $\text{g m}^{-2} \text{ cm}^{-1}$ ; BV=5  $\text{g m}^{-2} \text{ cm}^{-1}$ ; PP=6  $\text{g m}^{-2} \text{ cm}^{-1}$ ; JP=10  $\text{g m}^{-2} \text{ cm}^{-1}$ ). We should note here that sediment bulk densities ( $\text{g dry sediment cm}^{-3}$ ) were estimated from other locations in Chesapeake Bay.

Several possible mechanisms exist to explain this trend. Higher particulate nutrient concentrations in sediments may result because material

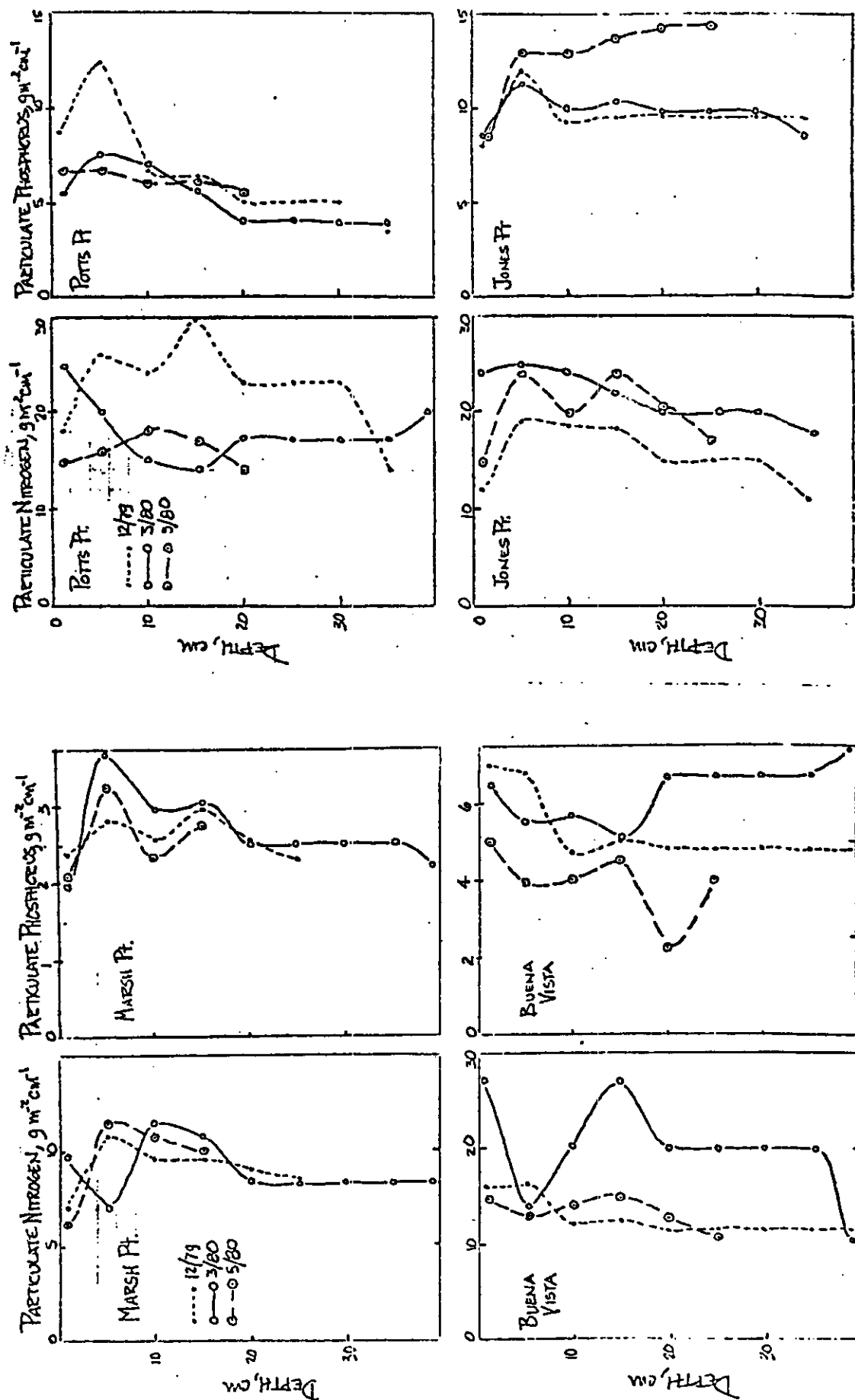


Fig. 8. Vertical profiles of particulate nitrogen and phosphorus from the Patuxent Estuary, 1979-1980. Data expressed as  $\text{gm}^{-2} \text{cm}^{-1}$ .

being deposited in the upper estuary was richer in these elements than in downriver locations or because the remobilization of these compounds was less complete in the upper estuary, perhaps due to rapid burial under conditions of high sedimentation. While our data base is sparse, there appeared to be a general decrease in water column seston:PN and seston:PP ratios in an upriver direction, suggesting that differential inputs may, in part, be responsible for differences in sediment stocks of PN and PP.

Secondly, there was a strong trend evident relative to the ratio of PN:PP in sediments wherein this ratio declined in an upriver direction. Ratios of PC:PN:PP indicated that the decline in the PN:PP ratio was caused by enrichment of PP rather than decreases in PN. It is probable that enrichment of PP in sediments resulted from adsorption of dissolved phosphorus compounds. Such processes have been shown to occur in the low salinity zone (Stirling and Wormald, 1977) of estuaries and thus this explanation is consistent with PP distributions observed in this study.

One obvious question that emerges from considering the storages of PN and PP in sediments concerns relationships between the amount of N and P entering the estuary and the amount stored in sediments on an annual basis. Boynton et al. (1982) have estimated that about  $10 \text{ g N m}^{-2} \text{ y}^{-1}$  and about  $1 \text{ g P m}^{-2} \text{ y}^{-1}$  enter the Patuxent estuary. Using a net sedimentation rate of about  $2 \text{ mm y}^{-1}$  (see latter sections of this report) and mean PN and PP values of  $10 \text{ g N m}^{-2} \text{ cm}^{-1}$  and  $2.5 \text{ g P m}^{-2} \text{ cm}^{-1}$  for sediments in the lower estuary (Fig. 8) we arrive at the conclusion that about 20% of the nitrogen and 50% of the phosphorus entering the estuary each year is buried. Thus, it appears that

burial of nitrogen and especially phosphorus is an important sink for these materials. The lack of substantial decreases in PN and PP with depth in the sediments further suggests that burial is indeed a relatively permanent sink for these compounds (Fig. 8).

#### Interstitial Nutrient Concentrations and Implications

Concentrations of several nutrient species ( $\text{NH}_4^+$  and DIP) in interstitial waters are depicted in Fig. 9 and presented in tabular form in Appendix 4. Ammonium concentrations ranged from 8 to 3130  $\mu\text{g-at l}^{-1}$  and, with few exceptions, increased steadily with depth in the sediments. Ammonium concentrations at the upriver station (Jones Pt.) were consistently higher, often by an order of magnitude, than at the remaining stations. In general, values reported here were lower than those observed in other areas of the Chesapeake Bay (Mattisoff et al., 1975).

Concentrations of DIP ranged from 1.0 to 257  $\mu\text{g-at l}^{-1}$  and generally increased with depth in the sediment. Concentrations were lowest at the upriver station (Jones Point) where concentrations of PP were highest. In all cases (i.e. stations and sampling times) dissolved forms of N and P were a very small fraction of the total N and P stored in the sediments.

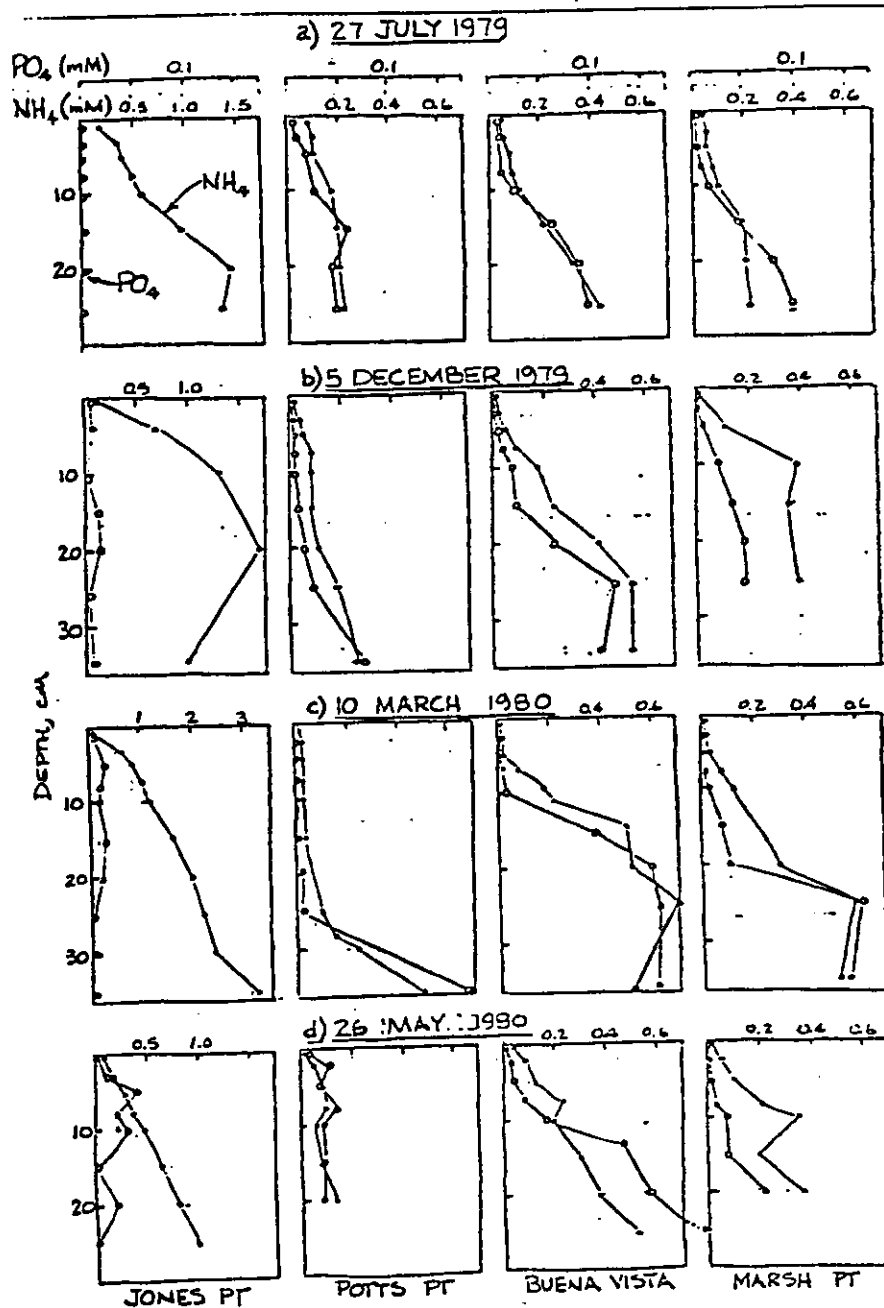


Fig. 9. Vertical profiles of interstitial water concentrations of ammonium and inorganic phosphorus at selected stations in the Patuxent River Estuary, 1979-80.

In recent years the flux of remineralized ammonium across the sediment/water interface is a process which has been intensively studied as indicated in the reviews of Zeitzschel (1979), Boynton et al. (1980a), and Nixon (1981). These authors have stressed the importance of this source of recycled  $\text{NH}_4^+$  for phytoplankton production. It has been generally assumed that the mechanism for generating such  $\text{NH}_4^+$  fluxes is the continual decomposition of organic matter deposited over decades, which leads to establishment of vertical concentration gradients (e.g. Berner 1976). Gradient-supported diffusive flux then accounts for mass transport across this interface, which can be calculated from solution of diagenic equations (Berner 1980). However, a number of investigators have recently compared  $\text{NH}_4^+$  fluxes calculated in this way from pore-water profiles of concentration with direct measurements of flux (obtained in-situ by change in concentration under bell jars or other benthic chambers).

The results of these studies indicated sporadic and generally poor agreement between the two estimates. We present a summary of these results in Fig. 10 which was extracted from Kemp et al. (1982). Data for Cape Lookout Bight are from Klump and Martens (1981), those for Narragansett Bay are from McCaffrey et al. (1980) and Elderfield et al. (1981), while the Chesapeake Bay data are taken from Callender (1979) and Callender and Hammond (1982) for the Potomac and this report for the Patuxent. Directly measured values tended to be greater than those calculated from concentration gradients, with ratios of the former to the latter ranging from about 1.5:1 to 15:1. Various authors have suggested that actual transport of pore-waters ions across the sediment/water interface can exceed that predicted from molecular diffusion

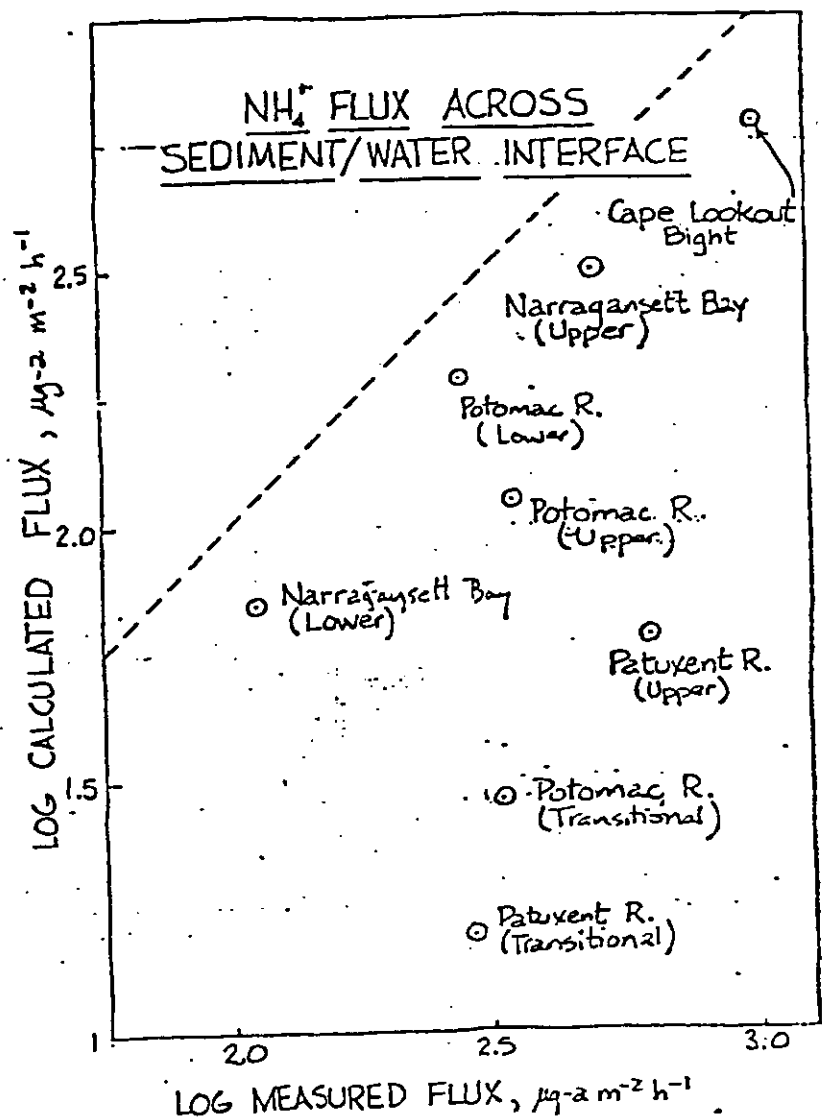


Fig. 10. Ammonium flux across the sediment/water interface as calculated from pore-water concentration gradients using a diffusion model versus in-situ measurements of flux.



because of one or more factors, including: stirring of sediments from currents or bioturbation; irrigation of sediments by suspension-feeding invertebrates; or ebulation from methane bubbles (e.g. Klump and Martens, 1981; Callender and Hammond 1982). Presumably, any or all of these mechanisms may be operative at any given time or place in a particular estuary. The extreme ratios in Potomac and Patuxent transitional reaches may be largely attributable (as suggested by Callender and Hammond, 1982) to bioturbation. Yet, there is by no means a one-to-one relation between this ratio and benthic infaunal densities in the Patuxent (compare this report to Holland and Hiegel, 1981).

We suggest another mechanism to explain this difference between calculated and observed fluxes--namely, that disproportionately high rates of organic decomposition occur at the uppermost sediment surface (about 0.5-2 mm, depending on deposition rates). These higher rates occur in the unconsolidated layer because the recently deposited organic material is far more labile than the refractory substances which remain at depth, and because oxygen is continuously supplied from the overlying water. Between this surface layer and the RDL\* more refractory material is decomposed aerobically, while below the RDL the most refractory material degrades slowly under anoxic conditions.

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\* redox discontinuity layer

While net uptake of  $\text{NH}_4^+$  may occur in certain seasons and places, vertical profiles of particulate composition in the Patuxent indicate that, overall, net regeneration is probably occurring in this floc layer. We present such profiles in %C, N and P in suspended and deposited particulate material for a water/sediment column in the spring at a lower Patuxent estuarine station (Fig. 11) and in detail for the sediments (Appendix 5). Both carbon and nitrogen exhibit continual decrease in % composition from 5 m above the sediment surface to 35 cm below it. However, there is a sharp discontinuity in % C and N across the sediment/water interface, suggesting rapid decomposition and remineralization occurring in the flocculent layer. Phosphorus, on the other hand, exhibits a local maximum % concentration at the sediment surface, indicating perhaps sorption and/or precipitation occurring in these oxidized upper sediments. Thus, we hypothesize that much of the metabolic activity which yields  $\text{NH}_4^+$  regeneration across the sediment/water boundary is, in fact, concentrated at that estuarine interface. While the gradient-supported estimates of  $\text{NH}_4^+$  flux may represent more the continued decomposition of organic matter accumulated over decades, direct measurements obtained in-situ with benthic chambers probably include the additional  $\text{NH}_4^+$  recycling of recently deposited organics which rest in the upper, floc layer.

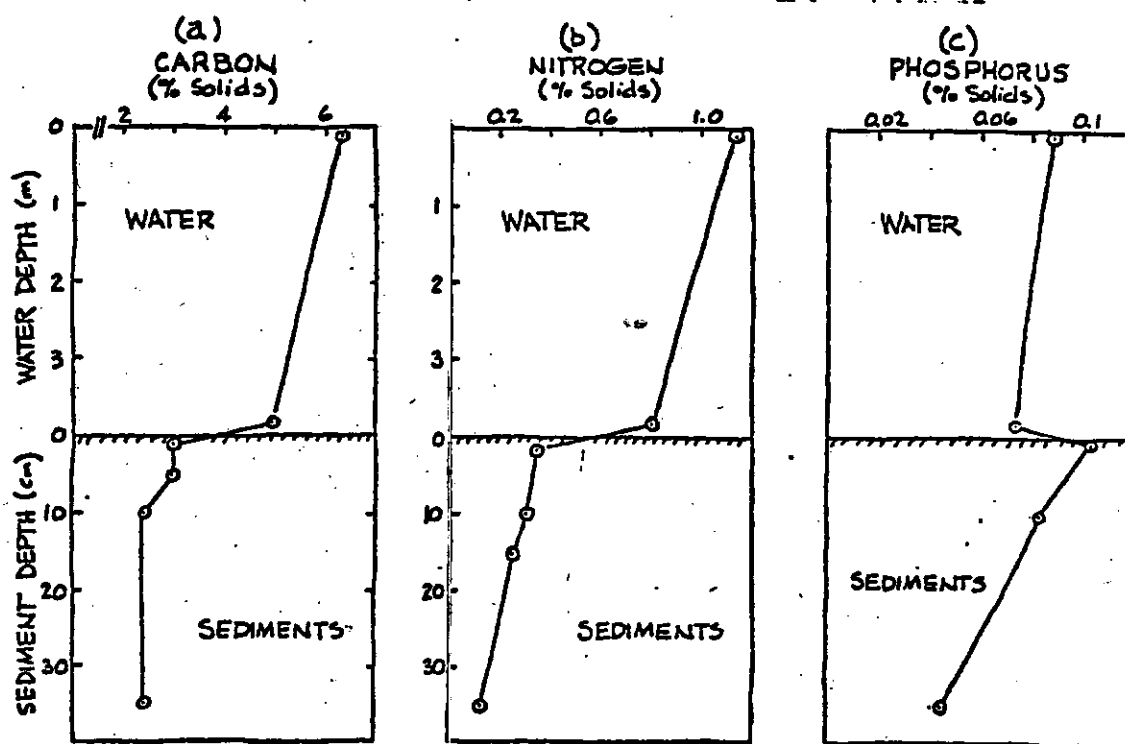


Fig. 11. Vertical profiles of particulate matter composition, as (a) carbon, (b) nitrogen, and (c) phosphorus, from 500 cm above sediment surface to 35 cm below. Data are for a station in the lower Patuxent River Estuary (Boynton et al. 1982).

### Sediment Nutrient Fluxes

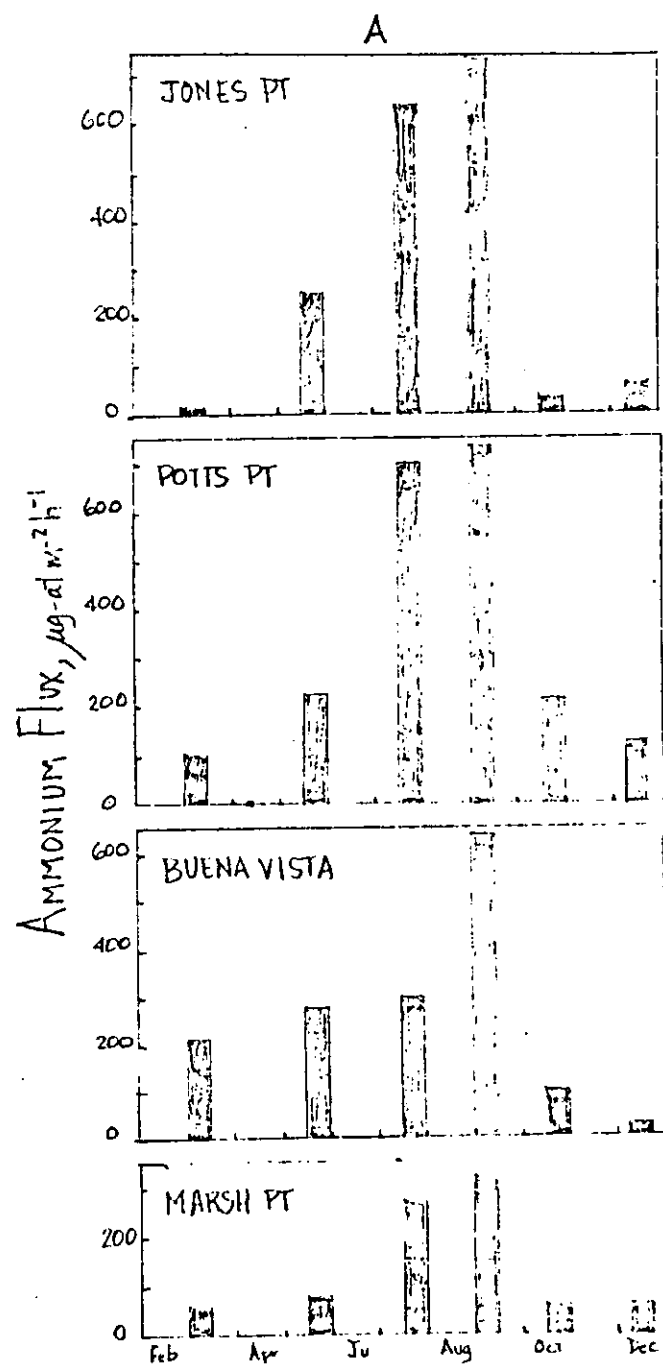
Net fluxes of several important nutrient species ( $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , DON,  $\text{NO}_2^-$ , DIP, DCP) between sediment and the water column were estimated at 4 locations on six occasions in 1979-1980. Selected flux data are shown in Fig. 12 and the complete data set is in Appendix 6.

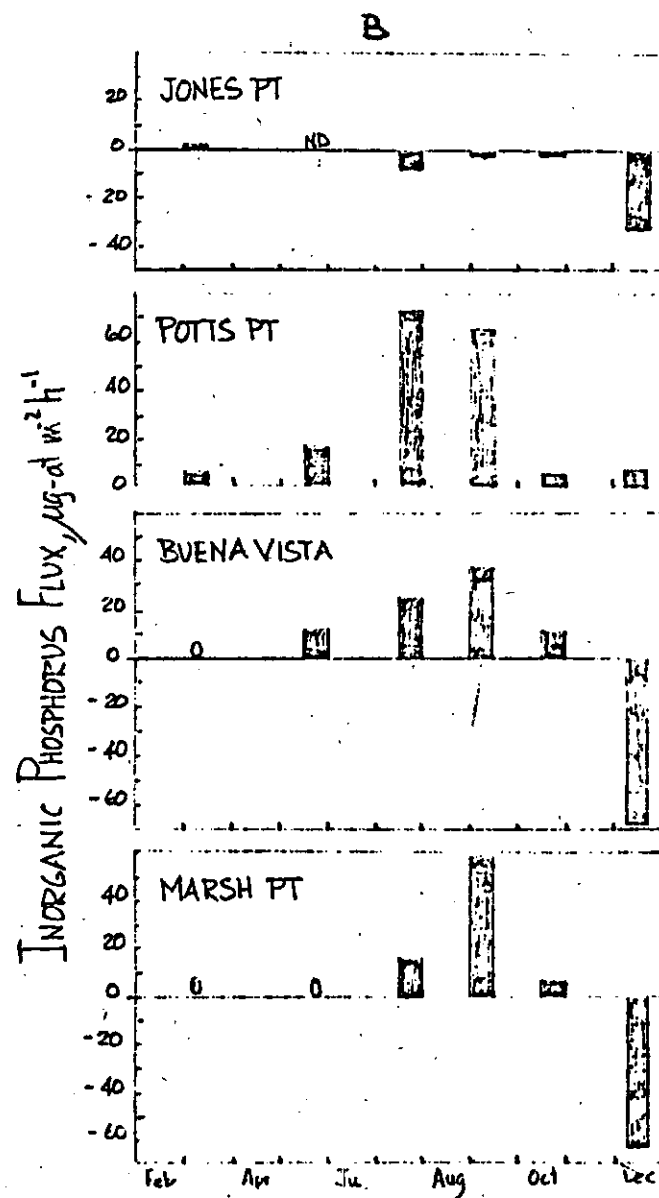
As we observed in an earlier study (Boynton et al., 1980a), reasonably clear seasonal and spatial patterns were also evident in this data set. In the case of ammonium, fluxes ranged from near zero to about  $700 \mu\text{g-at m}^{-2} \text{ hr}^{-1}$  and in almost all cases were directed out of the sediments (Fig. 12a). Maximum and minimum fluxes occurred in the summer and winter, respectively. In addition, the station downstream of the sediment trap (Marsh Point) generally had lower ammonium flux rates, while those in the turbidity maximum region had the highest. Time-weighted annual fluxes of  $\text{NH}_4^+$  were 15.4, 30.2, 38.2, and  $31.2 \text{ g N m}^{-2} \text{ y}^{-1}$  at Marsh Point, Buena Vista, Potts Point and Jones Point, respectively.

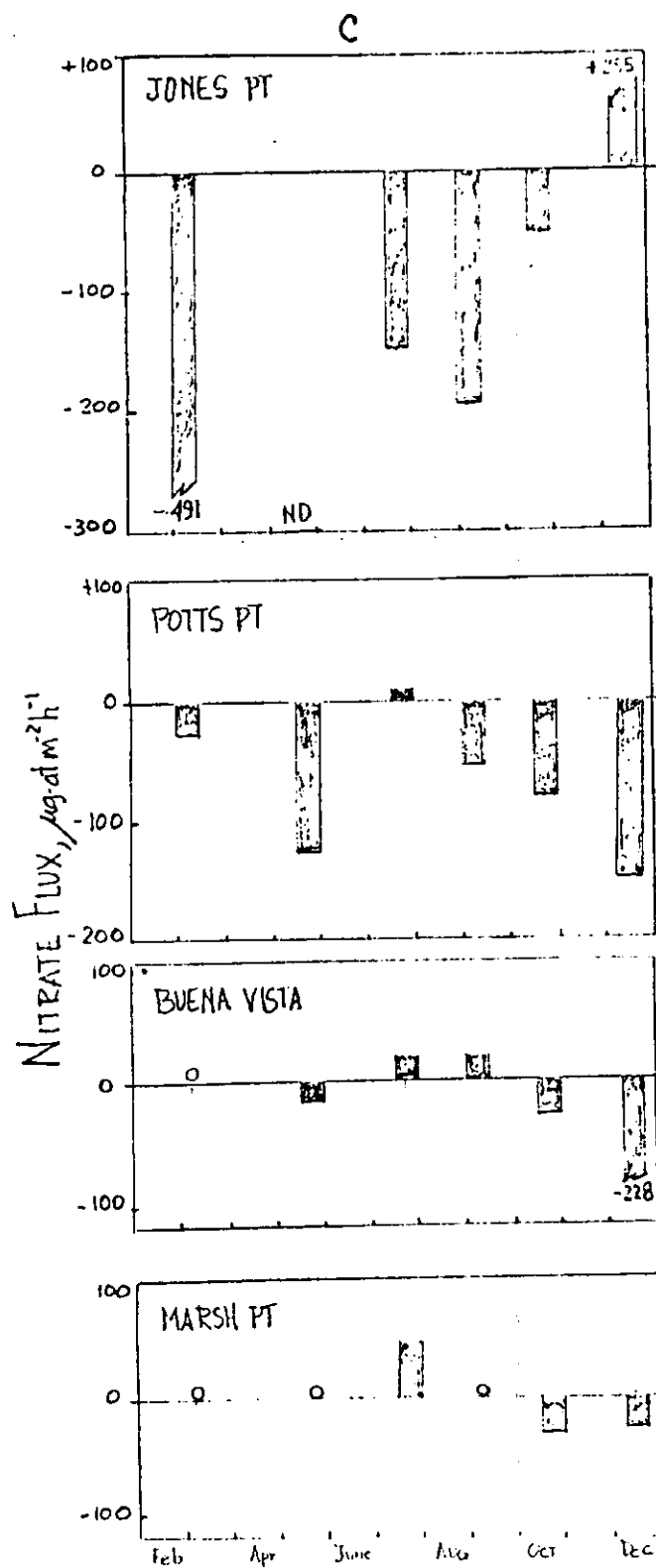
Net fluxes of DIP ranged from  $-60$  to  $70 \mu\text{g-at P m}^{-2} \text{ hr}^{-1}$  (Fig. 12b). Maximum and minimum rates occurred in the summer and winter, respectively, reflecting the same pattern as observed for ammonium flux. Consistently, DIP fluxes in the tidal-fresh region were the smallest observed during each survey and were often directed into the sediments. The two stations near the lower end of the turbidity maximum zone (Potts Point and Buena Vista) usually exhibited the highest release rates. Annual net fluxes (time-weighted) were 0.3, 0.5, 6.8 and  $-1.8 \text{ g P m}^{-2} \text{ y}^{-1}$  at Marsh Point, Buena Vista, Potts Point and Jones Point, respectively.

Fig. 12. Summary (mean values) of net sediment fluxes observed at 4 locations in the Patuxent estuary, 1979-1980. Zeros represent no net flux; ND signifies no data.

(A) Ammonium, (B) DIP, (C)  $\text{NO}_3$ , (D) DON, (E) Summary

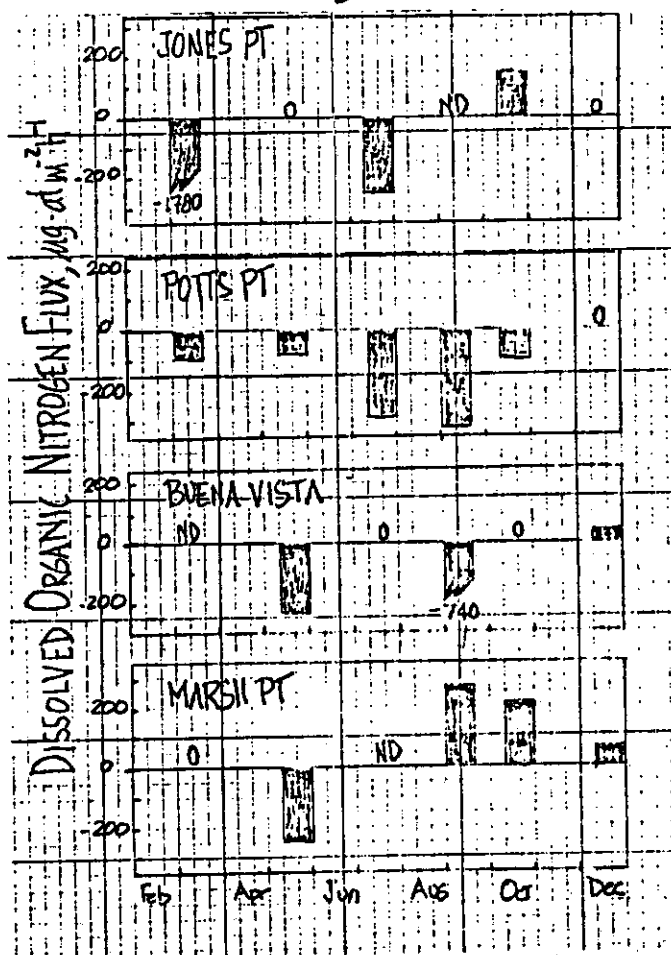


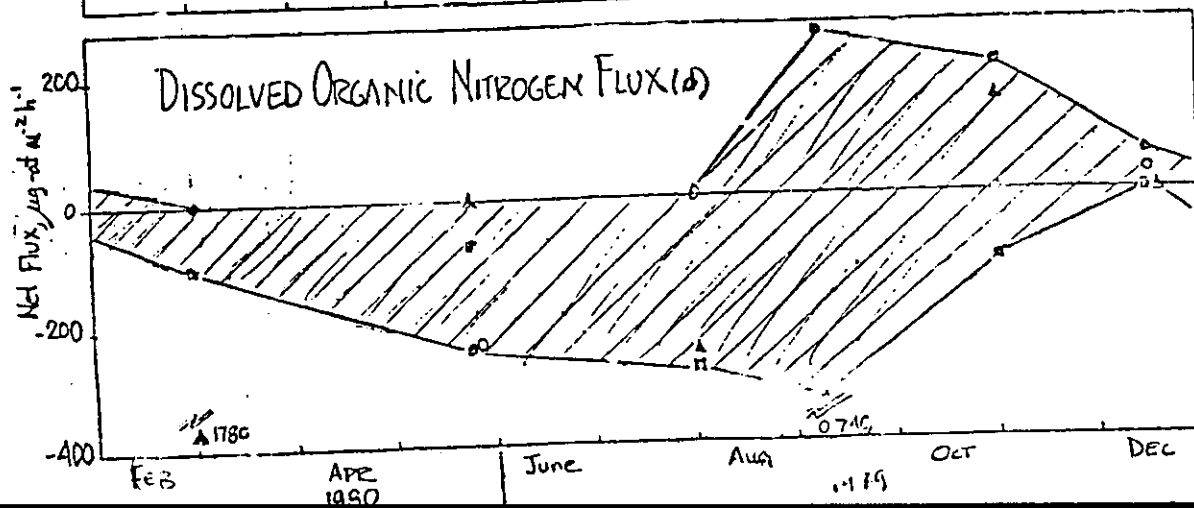
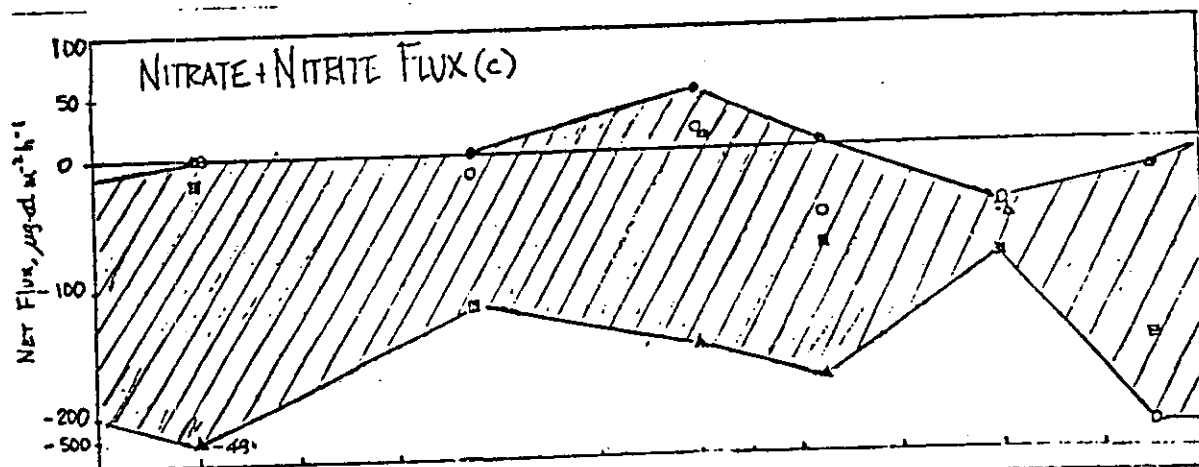
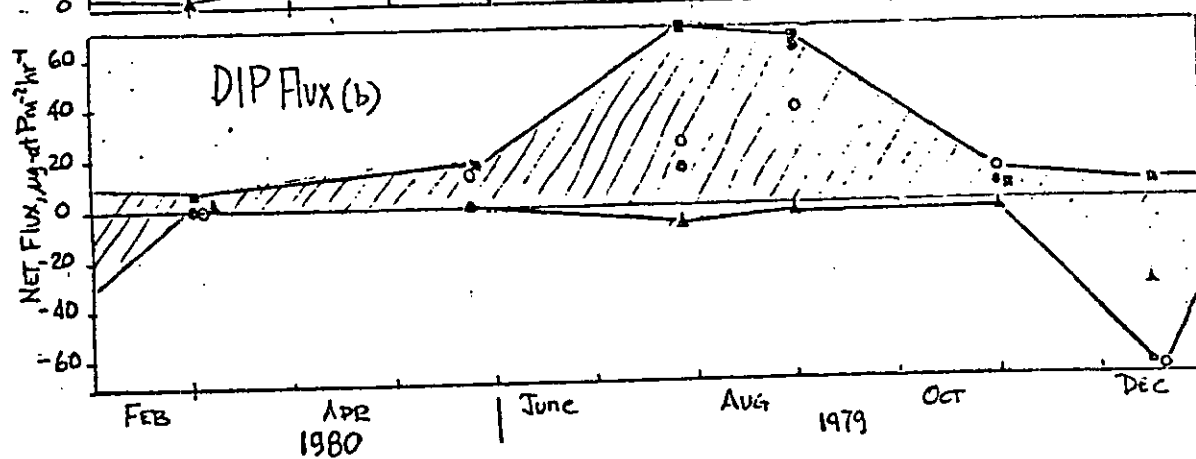
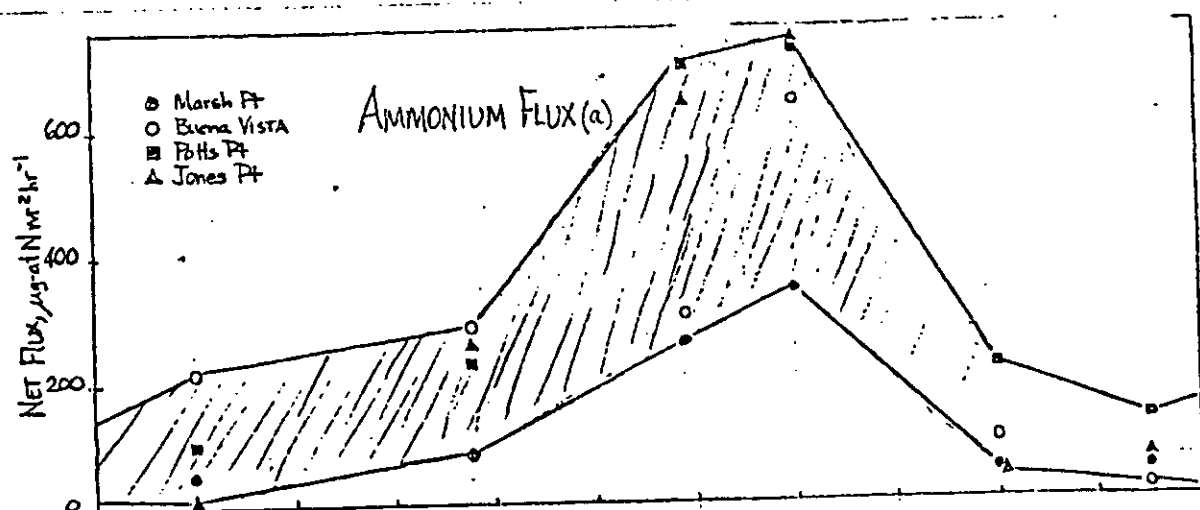






D





- The vast majority of nitrate and nitrite flux rates were either close to zero or were directed into the sediments (Fig. 12c). In an earlier study (in the Potts Point and Buena Vista area) data were analyzed using an annual average input-output model and it was inferred that nitrate + nitrite fluxes directed into the sediment may well have been entering a denitrification pathway (Boynton et al. 1980a). More recently, measurements by Jenkins (1981) using  $^{15}\text{N}$  techniques directly measured denitrification as well as nitrification rates. During the spring, both nitrification and denitrification were occurring in the sediments. Due to some uncertainty with the method, Jenkins did not report rates (in a strict sense) but rather referred to fluxes as denitrification or nitrification potential. In any case, activity was substantial in the spring and was consistent with earlier findings. In the present study, nitrate fluxes were highest in those portions of the estuary where  $\text{NO}_3^-$  was most abundant in the water column. For example, at the Buena Vista and Potts Point site, nitrate fluxes were close to zero during the summer and early fall when nitrite was not abundant in the water column, while fluxes were mainly directed into the sediment at these sites during winter and early spring when nitrate was more abundant. Consistent with the above observations, nitrite was often found entering the sediment at the Jones Point site where nitrate concentrations were elevated throughout most of the year. Annual nitrate fluxes were 0.6, 6.2, 9.2, and 14.5  $\text{g N m}^{-2} \text{ yr}^{-1}$  at Marsh Point, Buena Vista, Potts Point, and Jones Point, respectively.

Fluxes of dissolved organic nitrogen were somewhat erratic, but often large, and primarily directed into the sediments (Fig. 12d). This is in contrast to data collected in 1978-1979, which indicated DON fluxes to be

small and quite variable with regard to direction of the flux. In 1979-80, positive fluxes were observed only in the fall and early winter. Negative or zero fluxes dominated the remainder of the year.

Fluxes of dissolved organic phosphorus were, without exception, below detection level at all stations throughout the study period. This is basically consistent with earlier findings (Boynton et al. 1980a).

To obtain a view of the absolute magnitude and direction of all nitrogen fluxes between the sediment and the water column, several components of sediment-nutrient fluxes were summed and compared (Table 5). Fluxes were divided into two components, where  $N_1$  represented ammonium + DON, while  $N_2$  represented fluxes of  $NO_2$  and  $NO_3$ . This division was made primarily because nitrate + nitrite fluxes were almost always directed into the sediment, while ammonium and DON fluxes were primarily directed out of the sediments. The objective here was to obtain an overall view of the mass transfer of nitrogen between sediments and the water column (not including  $N_2$  or  $N_2O$  associated with denitrification). As indicated in Table 5, net fluxes of dissolved nitrogen species were directed from the sediments to the water column; there was a net release of dissolved nitrogen species from the sediments to the water column.

However, several additional points should be mentioned. The first is that nitrogen release from the sediments was most predominant during the warm periods of the year. With one exception (September, 1979, at Buena Vista) there were substantial releases of nitrogen from the sediments to the water column at all stations. In contrast, during the cooler seasons of the year,

Table 5. Comparison of different components of sediment nutrient fluxes at 4 locations in the Patuxent River, 1979-1980. N<sub>1</sub> and N<sub>2</sub> represent (NH<sub>4</sub> + DON) and (NO<sub>2</sub> + NO<sub>3</sub>) fluxes, respectively.

	Distance from River Mouth, Km							
	51		39		33		25	
	Jones Pt.		Potts Pt.		Buena Vista		Marsh Pt.	
	N <sub>1</sub>	N <sub>2</sub>	N <sub>1</sub>	N <sub>2</sub>	N <sub>1</sub>	N <sub>2</sub>	N <sub>1</sub>	N <sub>2</sub>
July, 1979	382	-145	409	8.2	302	18.8	272.6	42.5
Sept, 1979	744	-182	402	71.0	-100	-48	606	3.2
Oct., 1979	184	-53	109	-79	98	-44	263	-52
Dec., 1979	61	255	128	-150	57	-235	116	-27
Feb., 1980	-1770	-294	10	-26.0	215	0.0	53	0.0
May, 1981	253	3.4	150	-128	39	-16	-166	0.0

there were occasions when nitrogen fluxes were predominantly directed into the sediments, and, with one exception (February, 1980, Jones Point), the flux into the sediment was dominated by nitrate. This pattern of nitrogen release and uptake by the sediments is interesting to the extent that there is contained here an indication of estuarine homeostasis with regard to nutrient concentration in the overlying water. During the warmer periods of the year when nutrient fluxes from the sediment to the water column predominate, water column nutrient concentrations are low and hence the sediments serve as a source of nutrients to the water column. In contrast to this, during the colder parts of the year, water column concentrations of nutrients (and in particular nitrate) are very high (greater than  $100 \mu\text{g-at l}^{-1}$ ) and we see that there is substantial uptake of nutrients by the sediments, thus tending to decrease water column concentrations of nutrients. During the period of the year when water column concentrations are low, sediments appear to serve as a source which is important in maintaining plankton production, while during the cold season (when water column concentrations are high) sediments serve as a nutrient sink.

We have attempted to explore possible relationships between benthic nutrient fluxes and sediment oxygen demand with water column concentrations of various nutrient species, and such environmental variables as salinity, temperature, and several indices of water column production and respiration. The results of correlation analyses by station are given in Table 6. Fluxes of DIP did not reveal consistent relationships to any nutrient concentrations in the water column, but correlation coefficients were reasonably high between DIP flux and temperature at 3 stations (MP, BV, PP). If one flux measurement

Table 6 Results of correlation analysis relating selected benthic flux rates to water quality variables in the overlying water column. Stations are treated separately as indicated. An asterisk denotes a significant correlation ( $P < 0.05$ ).

INDEPENDENT VARIABLES										
<u>Marsh Pt.</u>										
<u>Fluxes</u>	<u>DIP</u>	<u>DON</u>	<u>NH<sub>4</sub>c</u>	<u>NO<sub>2</sub>c</u>	<u>NO<sub>3</sub></u>	<u>Salinity</u>	<u>Temp.</u>	<u>SOD</u>	<u>P<sub>a</sub></u>	<u>R<sub>u</sub></u>
DIP <sub>f</sub>	0.81*	0.21	0.74*	0.62	-0.81*	0.05	0.71	0.46	0.24	0.56
DON <sub>f</sub>	0.78	-0.40	0.29	0.84*	0.18	-0.33	0.23	0.48	-0.27	0.57
NH <sub>4</sub> <sub>f</sub>	0.87*	-0.47	0.74	0.44	-0.64	-0.32	0.84*	0.09	0.22	0.46
NO <sub>2</sub> <sub>f</sub>	0.15	-0.55	0.24	-0.37	-0.45	0.13	0.26	0.75*	0.29	-0.14
NO <sub>3</sub> <sub>f</sub>	0.22	-0.24	0.14	-0.36	-0.76*	0.14	0.58	-0.19	0.27	-0.33
SOD <sub>f</sub>	0.36	0.45	0.17	0.60	-0.13	-0.25	0.34	1.00	-0.11	0.31
<u>Buena Vista</u>										
<u>Fluxes</u>	<u>DIP</u>	<u>DON</u>	<u>NH<sub>4</sub></u>	<u>NO<sub>2</sub></u>	<u>NO<sub>3</sub></u>	<u>Salinity</u>	<u>Temp.</u>	<u>SOD</u>	<u>P<sub>a</sub></u>	<u>R<sub>u</sub></u>
DIP <sub>f</sub>	0.42	0.90*	-0.10	0.16	-0.93*	0.11	0.70	0.42	0.67	0.70*
DON <sub>f</sub>	-0.65	-0.35	-0.02	-0.34	0.51	-0.06	-0.56	-0.15	-0.59	-0.66
NH <sub>4</sub> <sub>f</sub>	0.65	0.51	-0.15	0.16	-0.73*	0.05	0.70	0.45	0.70*	0.32*
NO <sub>2</sub> <sub>f</sub>	-0.75	-0.03	-0.10	-0.66	0.18	0.03	-0.41	-0.29	-0.23	-0.44
NO <sub>3</sub> <sub>f</sub>	0.21	0.98*	-0.27	-0.01	-0.96*	0.30	0.54	0.38	0.53	0.24
SOD <sub>f</sub>	0.70	0.24	-0.62	0.05	-0.34	0.39	0.51	1.00	0.07	0.63
<u>Potts Pt.</u>										
<u>Fluxes</u>	<u>DIP</u>	<u>DON</u>	<u>NH<sub>4</sub></u>	<u>NO<sub>2</sub></u>	<u>NO<sub>3</sub></u>	<u>Salinity</u>	<u>Temp.</u>	<u>SOD</u>	<u>P<sub>a</sub></u>	<u>R<sub>u</sub></u>
DIP <sub>f</sub>	0.87*	0.52	-0.32	-0.05	-0.90*	-0.01	0.89*	0.57	0.46	0.15
DON <sub>f</sub>	-0.79*	-0.77	0.40	-0.17	0.92*	-0.04	-0.84*	-0.60	-0.45	-0.40
NH <sub>4</sub> <sub>f</sub>	0.91*	0.61	-0.26	0.01	-0.93*	-0.12	0.91*	0.65	0.77	0.19
NO <sub>2</sub> <sub>f</sub>	-0.71	-0.67	-0.28	-0.72	0.52	0.37	-0.49	-0.76*	0.33	-0.61
NO <sub>3</sub> <sub>f</sub>	0.19	0.53	-0.87	-0.30	-0.56	0.69	0.37	-0.07	0.06	0.42
SOD <sub>f</sub>	0.74*	0.69	0.22	0.68	-0.72	-0.59	0.76*	1.00	0.80*	0.34
<u>Jones Pt.</u>										
<u>Fluxes</u>	<u>DIP</u>	<u>DONc</u>	<u>NH<sub>4</sub></u>	<u>NO<sub>2</sub></u>	<u>NO<sub>3</sub></u>	<u>Salinity</u>	<u>Temp.</u>	<u>SOD</u>	<u>P<sub>a</sub></u>	<u>R<sub>u</sub></u>
DIP <sub>f</sub>	-0.00	-0.45	-0.17	0.55	-0.26	----	0.28	0.18	0.17	0.02
DON <sub>f</sub>	-0.62	0.35	-0.64	-0.18	-0.31	----	0.16	-0.21	0.14	-0.71
NH <sub>4</sub> <sub>f</sub>	-0.51	0.77	-0.60	0.78*	-0.84*	----	0.92*	0.96*	0.49	0.60
NO <sub>2</sub> <sub>f</sub>	-0.58	0.93*	-0.68	0.74*	-0.66	----	0.66	0.37	0.66	0.35
NO <sub>3</sub> <sub>f</sub>	-0.38	0.36	-0.26	-0.46	-0.03	----	-0.05	-0.09	-0.10	-0.22
SOD <sub>f</sub>	-0.39	0.56	-0.47	0.61	-0.73*	----	0.83*	1.00	0.26	0.59

is deleted from the data base at Buena Vista, then significant temperature relationships exist at the three down-river stations.

Correlation analyses relating DON flux to a variety of independent variables did not reveal any consistent significant correlations reflecting the erratic nature of these fluxes. At present, dissolved organic nitrogen is treated as if it were a defined form of nitrogen; in fact, it represents unknown and probably complex mixes of dissolved organic compounds. As defined, we find no seasonal or consistent patterns in these fluxes, and hence no significant correlations. Understandable patterns might emerge if more effort were expended in better defining the specific components which comprise the dissolved organic nitrogen fraction. It is worrisome that a nutrient species about which we know so little can at times be a substantial component of the nutrient exchange across the sediment-water interface.

In a similar fashion, ammonium fluxes were not consistently and significantly correlated with water column nutrient variables. However, there was a reasonably good relationship to temperature suggesting that a major portion of the ammonium flux was dependent on remineralization by bacteria and other small heterotrophs which are known to have a strong metabolic dependency on temperature. Nitrite fluxes, as indicated previously, were mostly small and somewhat variable, and because of this, no consistent significant correlations emerged between water column variables and nitrite fluxes. In contrast to nitrite, nitrate fluxes showed a strong relationship between nitrate flux and nitrate concentration in the water column at stations where the water column nitrate varied seasonally (MP and BV). The strong



correlation of nitrate concentration with the negative flux of nitrate suggests a nitrate consumptive process in the sediments which in some seasons may be limited by diffusion across the sediment-water interface. It apparently is not closely linked to aerobic respiration. These relationships suggest that nitrate fluxes into the sediment represent the first stage of either denitrification ( $\text{NO}_3^- \longrightarrow \text{N}_2\text{O}$  and  $\text{N}_2$ ) or nitrate respiration ( $\text{NO}_3^- \longrightarrow \text{NH}_4^+$ ). Van Kessel (1977) has shown that denitrification rates are dependent on  $\text{NO}_3^-$  concentration in the overlying water. This coupled with the fact that we did not observe large  $\text{NH}_4^+$  fluxes out of the sediments when  $\text{NO}_3^-$  was entering sediments strongly suggests that this  $\text{NO}_3^-$  uptake from water to sediments was associated with denitrification. Due to a few erratic measurements, which we nonetheless had no reason to discard, significant relationships between nitrogen concentration in the water column and nitrate flux did not emerge for all stations.

Somewhat surprisingly, we did not see consistent and significant relationships between SOD and such independent variables as temperature and water column metabolism. At two stations SOD and temperature were strongly related, as we have previously reported for other portions of the estuary. However, at the two most down-river stations, such a relationship did not emerge in the 1979-1980 data. In addition, no significant relationships emerged relating sediment oxygen demand with primary production in the water column. Nixon (1979) and Hargrave (1973) reported that strong relationships existed between organic matter production in the water column and sediment oxygen demand, at least on an annual basis, for a variety of ecosystems. In fact, on an annual basis our data fit this relationship reasonably well.

However, over shorter time periods (days) it is apparent that sediment oxygen demand is not dependent on the simultaneous rate of in-situ primary production.

When we initially began exploring potential relationships between a variety of independent variables and various sediment-water fluxes, we conducted correlation analyses on the entire data base (all stations were not treated individually as in Table 6). Interestingly enough, there were very few significant correlations which emerged from this aggregated data base relative to those which emerged from the analysis presented in Table 6. Based on previous work at two stations, we had anticipated that we would find significant relationships between such factors as nutrient flux, temperature, SOD, and primary production. The reduction in significant relationships found in the aggregated data base and the few consistent trends found in the station-by-station analysis suggests that there are, indeed, a variety of mechanisms regulating nutrient flux and that different mechanisms may well be operating contemporaneously in the same estuarine system. Results of correlation analyses conducted on the entire data base are given in Appendix 7.

We can make a preliminary assessment as to the relative importance of benthic releases of nitrogen and phosphorus by comparing these fluxes to the nutrient demands of phytoplankton production in the water column (Table 7). Previously in the Patuxent (Buena Vista and Potts Point), we have found that benthic fluxes of ammonium and DIP can satisfy from 0 - 190% and 52 - 330% of the estimated daily demand of these nutrients, respectively. The atomic

TABLE 7. ESTIMATES OF THE PERCENT OF NITROGEN AND PHOSPHORUS DEMAND OF PHYTOPLANKTON (Dp) WHICH ARE SATISFIED VIA AMMONIUM RELEASES FROM ESTUARINE SEDIMENTS (Sn)

DATE		Marsh Point			Buena Vista			Potts Point			Jones Point		
		Dp	Sn	%	Dp	Sn	%	Dp	Sn	%	Dp	Sn	%
7/27/79	N	17.34	6.41	37	17.3	7.13	41	20.3	16.63	82	21.0	15.29	73
	P	1.09	0.38	35	1.08	0.60	57	1.27	1.76	139	1.31	-----	--
9/5/79	N	24.4	8.00	33	19.7	15.36	78	25.8	17.52	68	12.4	17.76	143
	P	1.52	1.49	98	1.23	0.91	74	1.61	1.56	97	0.78	-----	--
10/31/79	N	7.3	1.34	18	4.21	2.35	56	7.10	5.06	71	1.9	0.89	45
	P	0.46	0.15	32	0.26	0.28	108	0.44	0.12	27	0.12	---	--
12/12/79	N	7.64	1.27	17	0.17	0.50	294	0.95	3.07	323	1.32	1.46	111
	P	0.48	---	--	0.01	---	---	0.06	0.17	283	0.08	---	--
2/28/80	N	3.1	1.27	41	1.8	5.16	287	8.0	2.47	31	1.1	0.24	22
	P	0.19	0	0	0.11	0	0	0.50	0.12	24	0.07	0.05	71
5/23/80	N	48.3	1.87	4	24.2	6.60	27	37.6	5.40	14	33.9	6.07	18
	P	3.02	0	0	1.51	0.29	19	2.35	0.39	17	2.13	---	--

ratios of O:N:P observed in plankton were used to estimate nutrient uptake. We have summarized similar calculations which involved the same assumptions for data collected in 1979-80 for four stations in the Patuxent River including Buena Vista and Potts Point. At the Buena Vista and Potts Point station, the percent of phytoplankton nitrogen demand supplied via ammonium fluxes from estuarine sediments ranged from about 19 - 323% of estimated daily demand, while the percent of phosphorus was supplied by the sediments ranged from 0 - 283% of estimated demand. During the most productive periods of the year (May through October) benthic fluxes of ammonium and DIP consistently are an important source of nutrient material, and at times a dominating source.

For comparative purposes we have summarized ammonium and phosphate fluxes for a variety of estuarine ecosystems (Fig. 13). Perhaps one of the most outstanding aspects of this figure is that sediment fluxes from the Patuxent are among the highest reported. For instance, fluxes of ammonium and nitrate were on the order of 325 and 50  $\mu\text{g-at m}^{-2} \text{ hr}^{-1}$ , respectively in 1978-1980. Fluxes of ammonium and DIP in Narragansett Bay were estimated to be 120 and 20  $\mu\text{g-at m}^{-2} \text{ hr}^{-1}$ , respectively, whereas in Buzzards Bay these fluxes were somewhat lower.

The high flux rates we observed may well be related to seasonal organic matter supplied to the benthos and physical dynamics of the sediment-water interface. Inputs of organic matter appear to be primarily derived from phytoplankton production during most of the year, although upland runoff and export from adjacent marshlands during winter and early spring may constitute a significant contribution.

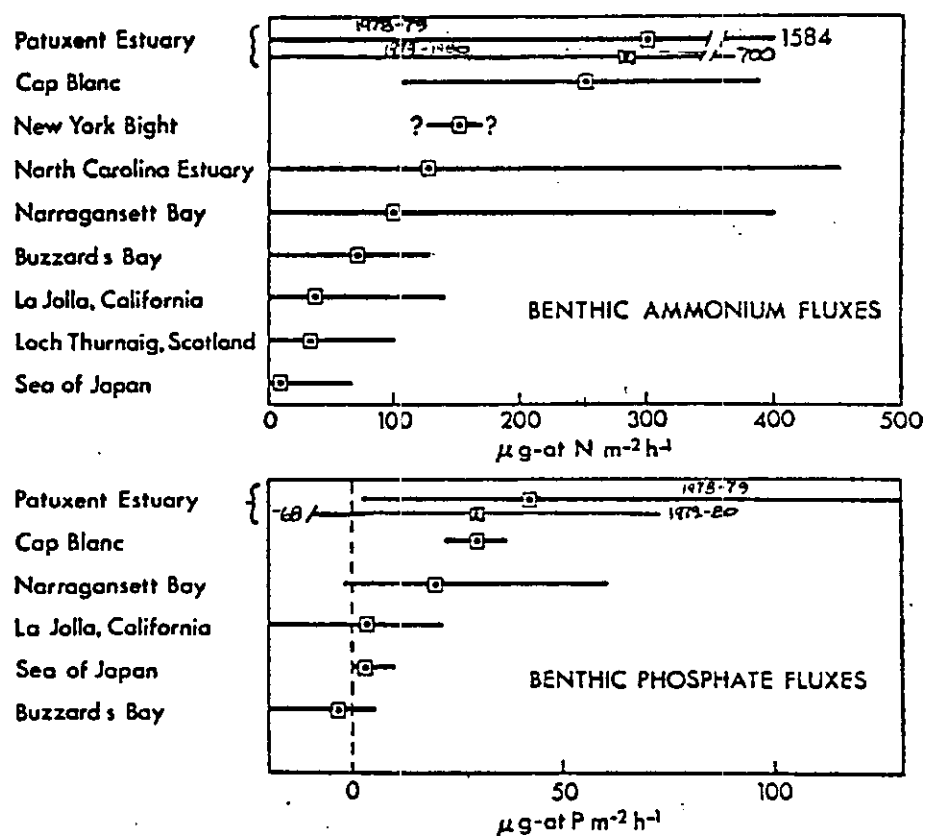


Fig. 13. Comparative benthic fluxes (annual mean and range) of  $\text{NH}_4^+$  and dissolved inorganic phosphate from selected estuarine and coastal ecosystems. Data collected in 1979-80 in the Patuxent are compared to those collected in 1978-79 at the Potts Point and Buena Vista stations. References are given in Boynton et al. (1980).

Analyses reported by Hargrave (1973 ) and Nixon (1979) indicate that benthic respiration, and perhaps nutrient flux, was related to organic matter delivery from the water column to the benthos. Our data fit their regressions reasonably well. However, as we noted earlier, correlations between concurrent measurements of sediment nutrient flux, benthic respiration and productivity in the water column were not strong. We conclude from this observation that there may be time lags between phytoplankton productivity and SOD and nutrient flux induced by short-term temporal variations and the rate of settling of organic matter to the bottom.

The second feature which contributes to high flux rates concerns the dynamic behavior of the sediment surface. As we shall see later in this report, results from sediment trap studies suggest very high rates of sedimentation and resuspension occur, particularly at the lower boundary of the turbidity maximum zone (Buena Vista and Potts Point). Moreover, it appears that sediments are extensively mixed via the burrowing activities of benthic infauna (Holland and Hiegel, 1981). The combination of these processes tends to enhance the exchange of dissolved materials between the sediment and the water column.

#### Deposition and Resuspension of Particulate Nitrogen and Phosphorus

During 1979-80, studies were conducted to obtain estimates of the deposition of inorganic particulates and particulate carbon, nitrogen and phosphorus on the sediment surface. This effort was undertaken to quantify

these fluxes and thus clarify relationships between SOD and nutrient fluxes which had previously been inferred only from correlation analysis (i.e. Boynton et al., 1980).

This effort was intensive, a large amount of data was collected regarding the deposition and resuspension of inorganic sediment, particulate carbon, nitrogen and phosphorus; but interpretation of these data is difficult for a number of reasons. One of the basic characteristics of many estuarine ecosystems, the Patuxent included, is the shallow depth. Because of this, surficial sediments are prone to resuspension, transport, and eventual resettling. This, in itself, would not pose a problem relative to the interpretation of sediment trap data except in areas where the resuspension phenomenon extends through an appreciable portion of the water column. If resuspension occurs throughout the water column, then it is impossible to partition deposition of new particulate material to the bottom from deposition of recently resuspended material. This did not occur at all stations at all times, but on occasion, it appears to have been a problem. Thus, in analyzing the data base, some subjective judgments had to be made concerning periods of time when we felt data indicated that resuspension was occurring throughout the water column. In those situations, data were not used in the calculation of net sedimentation.

A second problem involved the growth of fouling organisms on the collecting cups attached to the sediments traps strings. We anticipated that fouling would be a characteristic feature of this system, and hence there would be a need to include a series of blank cups on each sediment trap

string. However, during the course of the study we found evidence of a strong, vertical gradient in the rate of fouling. Specifically, fouling appeared to be far more intense in the near surface waters than in deeper waters, and far more intensive during the summer than during the winter periods. It would appear that in order to more accurately subtract the fouling effect from deposition, many more blank cups would be needed on each sediment trap string.

A final problem which is inherent in the sediment trap methodology concerns the degree to which organic materials occurring in the collection cups during the collection periods were metabolized. While there does not appear to be any way to rigorously analyze the amount of organic material respired in the collection cup over the collecting period, we attempted to minimize the effect of this error by deploying sediment cups for the minimum amount of time necessary to obtain measureable amounts of material. This problem does not apply to the use of sediment traps when used for determination of the vertical flux of inorganic particles, but certainly pertains when the vertical flux of organic compounds is of interest.

The results are recorded in Table 8 and include the range of estimates derived from 3 slightly different computation schemes. Unfortunately, the range is large. Deposition rates of N and P ranged from 25.7 to 228.9 g N m<sup>-2</sup> yr<sup>-1</sup> and from 5.8 to 96.0 g P m<sup>-2</sup> yr<sup>-1</sup>, respectively. Rates were consistently lower in the deeper, downriver stations, possibly because of less interference from resuspension and because of the longer water column, which would allow for utilization of a larger fraction of the organic matter prior to its reaching the bottom. Deposition rates were generally highest in the Buena Vista-Potts Point area of the estuary, consistent with the notion that



TABLE 8. ANNUAL ESTIMATES OF DEPOSITION OF PARTICULATE NITROGEN, PHOSPHORUS AND SEDIMENTS IN THE PATUXENT RIVER ESTUARY, DERIVED FROM SEDIMENT TRAPS.<sup>a</sup>

STATION	$\text{gm}^{-2}\text{yr}^{-1}$		
	Nitrogen	Phosphorus	Sediments
St. Leonard Creek	45.8 <sup>b</sup>	9.8	6,975
	35.5	7.08	5,075
	31.4	5.8	5,012
Marsh Point	61.6	16.4	8,194
	40.4	11.8	7,798
	25.7	8.3	5,583
Buena Vista (channel)	54.2	25.9	3,949
	39.7	19.8	10,096
	111.7	53.6	23,548
Buena Vista (shoal)	123.9	41.6	32,547
	128.2	40.7	31,579
	228.9	96.0	58,959
Potts Point	59.5	25.7	14,147
	39.2	24.6	13,287
	121.8	47.0	28,284
Jones Point	36.9	17.2	7,497
	31.1	14.2	6,326
	47.08	33.6	13,013

<sup>a</sup>Sediment traps were deployed on 10 occasions between August 1979 and July 1980. Deployment durations varied from 7 to 33 days depending on season.

<sup>b</sup>Triple entries represent results of 3 different calculations of deposition. See Appendix 8 for details.

deposition is greatest at the downstream edge of the turbidity zone. These rates may, for the reasons discussed earlier, be overestimates of net sedimentation, and thus this must be considered, at least qualitatively, when examining these values.

Net deposition rates of N and P were also determined using sedimentation rate estimates derived from inspection of pollen grain sequences (Brush et al. 1981). The rates of N and P deposition presented in Table 9 are somewhat comparable to those obtained from sediment traps but are considerably lower for sediments for reasons which are presently unclear.

Annual estimates of resuspension rates of particulate nitrogen and phosphorus are summarized in Table 10. Several interesting features are apparent in this table, the foremost being that estimates of resuspension are far larger than deposition rates, the difference being an order of magnitude or more. Secondly, resuspension rates were relatively low in the lower estuary where water depths were greater (10m), the effect of wave-induced resuspension negligible in open waters and tidal height change per unit of water depth was also minimal. Maximum rates occurred in the portion of the estuary where the water was shallow and where tidal height change per unit depth was greatest (Jones Pt.).

It should be noted that resuspension of P at Jones Point was very high relative to other sections of estuary and relative to the amount of nitrogen deposited at that site. Both of these observations are consistent with the N and P content of bottom sediments in that zone of the estuary, indicating that our measurements represent local resuspension phenomena.

TABEL 9. INDIRECT ESTIMATES OF NITROGEN, PHOSPHORUS AND SEDIMENT DEPOSITION IN THE PATUXENT RIVER ESTUARY.

LOCATION	Sedimentation Rate, mm <sup>1</sup>	Sediment Bulk Density g/cm <sup>3</sup>	Nitrogen	DEPOSITION <sup>2</sup>		Sediments
				Phosphorus	g m <sup>-2</sup> yr <sup>-1</sup>	
Lower Estuary (Marsh Pt. - St. Leonards Cr.)	1 - 3 mm ( $\bar{x}$ = 2.0)	1.5	32.0	6.3		3,000
Upper Estuary (Buena Vista - Potts Pt.)	4 - 7 mm ( $\bar{x}$ = 5.0)	1.5	80.3	15.8		7,500

<sup>1</sup> From Brush, et al (1981) based on analysis of pollen distributions in sediment coves.

<sup>2</sup> Calculated by multiplying sediments deposited per year by ratio of N: Sediments and P: Sediments observed in top several cups of sediment traps.  
N:Sediment = 0.0107 ± 0.0092; P:Sediment = 0.0021 ± 0.0012.

To evaluate possible mechanisms regulating the resuspension rate, we calculated annual average values for two variables which could have an influence, those being tidal rise and wind velocity (creating wave-induced turbulence). These are also shown in Table 10. Linear regression analysis indicated very strong and interpretable relationships existed between resuspension of N and P and our index of tidal energy. For these variables,  $r^2$  values were 0.97 and 0.82 for N and P, respectively. Regressions of N and P resuspension related to wind energy were also significant but slopes were negative. We can't envision a mechanism whereby lower wind (wave) action would create higher rates of resuspension and hence we view this relationship as spurious. Thus, from our preliminary analyses, it appears that tidal energy is an important factor regulating resuspension rates through a broad reach of the Patuxent estuary.

## ORGANIC CARBON DYNAMICS

### Plankton Production and Respiration

Plankton production and respiration were measured at four locations in the Patuxent River estuary on six different occasions between July, 1979 and May, 1980. In Figure 14 we provide plots of volumetric production ( $P_a$ ), each measurement plotted at the incubation depth. The integrated values of daytime net photosynthesis ( $P_a$ ) and respiration during hours of darkness ( $R_n$ ) are indicated on the diagram, as is the depth to which 1% of the surface insolation penetrated on the day of the measurement. Several things were apparent in this diagram. The euphotic zone depth generally increased in a

TABLE 10. ANNUAL ESTIMATES OF RESUSPENSION RATES OF PARTICULATE NITROGEN AND PHOSPHORUS IN THE PATUXENT RIVER ESTUARY DERIVED FROM SEDIMENT TRAPS.<sup>a</sup>

STATION	Resuspension Rates ( $\text{gm m}^{-2} \text{yr}^{-1}$ )		Tidal Energy <sup>b</sup> ( $\text{ftd}^{-1} \text{ft}^{-1}$ )	Wind Energy <sup>c</sup> ( $\text{Cm}^2 \text{S}^{-2}$ )
	Nitrogen	Phosphorus		
St. Leonards Cr.	142.1	29.3	0.072	180
Marsh Point	278.7	75.3	0.092	180
Buena Vista channel	1063.3	607.9	0.170	98
Buena Vista shoal	404.9 (733.9)	217.5 (412.7)	0.255 (0.213)	98 (98)
Potts Point	1021.6	517.0	0.235	98
Jones Point	945.6	304.4	0.743	31

<sup>a</sup>Sediment traps were deployed on 10 occasions between August 1979 and July 1980. Deployment durations varied from 7 to 33 days depending on season.

<sup>b</sup>Calculated as mean tidal range per day per depth.

<sup>c</sup>Calculated as wind velocity squared, including only winds with >1 Km fetch.

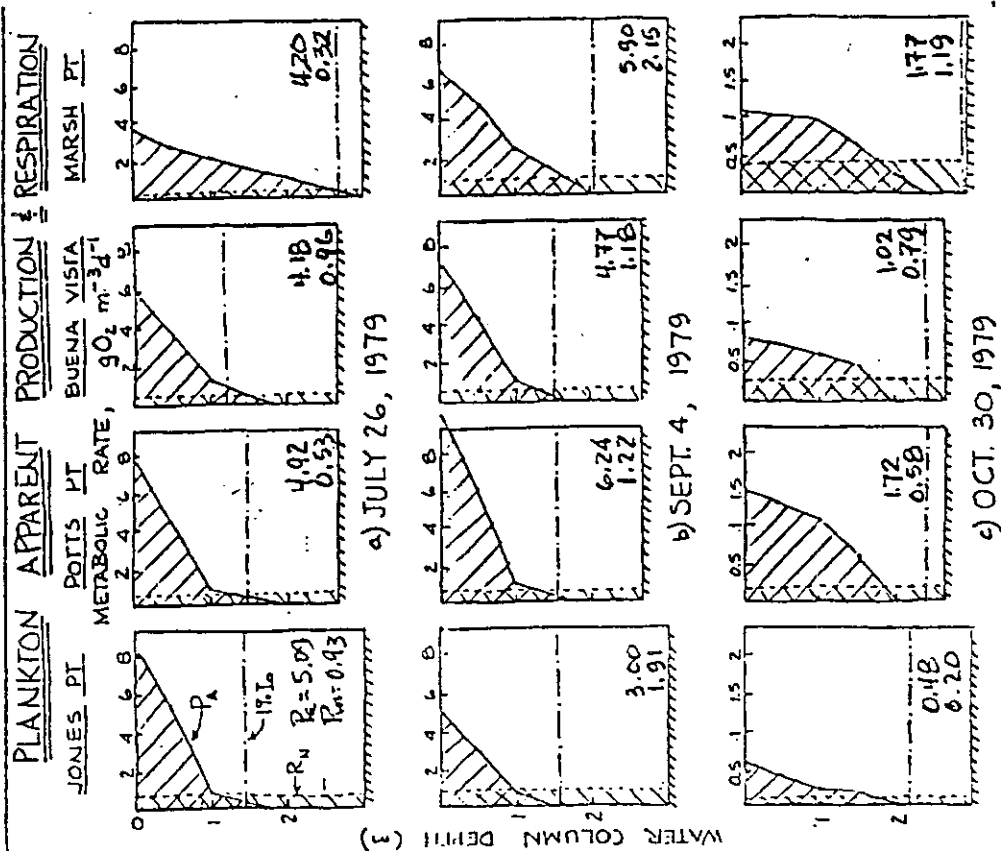
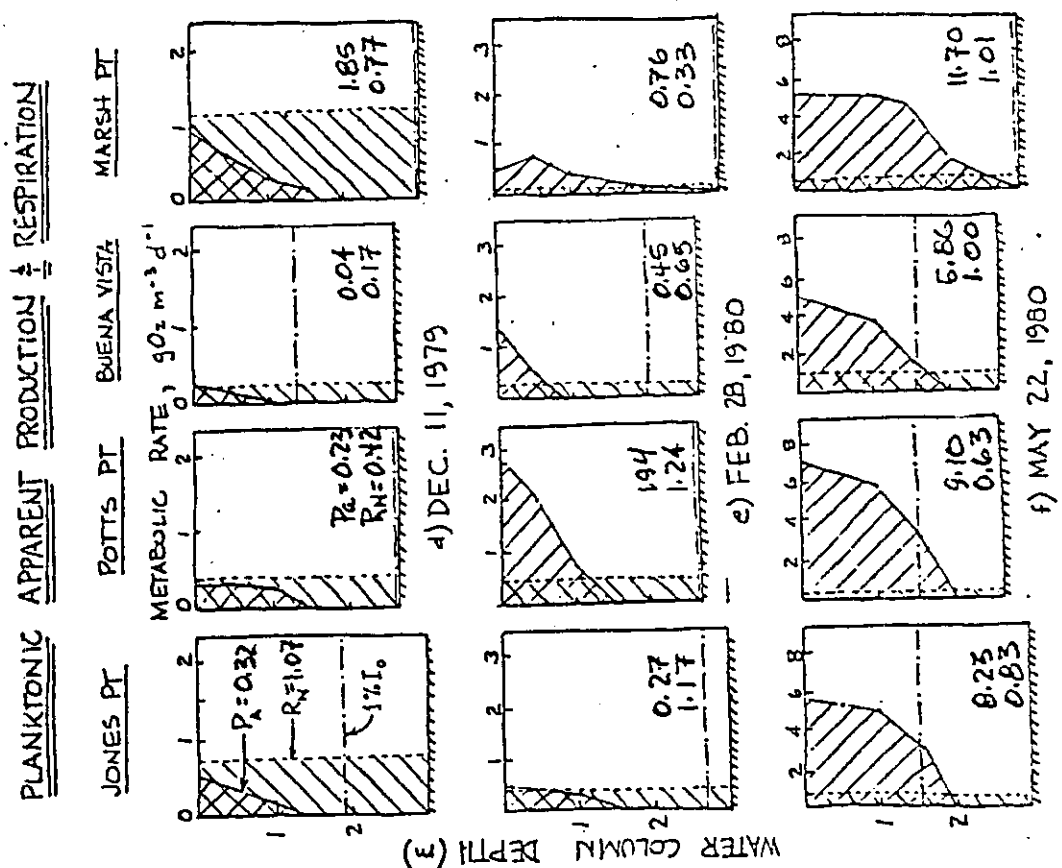


Fig. 14. Plankton apparent production ( $P_a$ ) and respiration ( $R_n$ ) at four stations in the Patuxent River Estuary, 1979-1980.

downstream direction, reflecting the transition from the more turbid areas which extend from Benedict (River mile 20) upriver, and the downstream mesohaline areas which are clearer. Secondly, volumetric rates of primary production show an inverse relationship to euphotic zone depth. On most occasions maximum volumetric productivity was observed in surface water incubations at the upriver stations. Integrated rates of  $P_a$  were, for the most part, comparable between stations. Throughout the study period, nutrient concentrations in the water ( $\text{NH}_4^+$ ,  $\text{NO}_3^-$ ,  $\text{PO}_4$ ) were less at the downriver station than at upriver locations, but the euphotic zone was deeper at the downriver station. Thus, it appears that light may be a factor limiting  $P_a$  in the turbidity maximum region. In the clearer portion of the estuary, nutrients may exert a limiting effect on volumetric rates of production, but the greater light availability compensates for nutrient limitation, and areal rates are similar throughout the study area. We should also note that  $P_a$  values were often high at PP (discharge canal station); on 3 of the six cruises, values at PP exceeded those at the downriver station. Tabular values of  $P_a$  and  $R_n$  are given in Appendix 11.

Seasonal patterns of plankton production and respiration are shown graphically in Figure 15. Daily average rates of  $P_a$  were calculated to be 4.4, 2.7, 4.0, and 2.9  $\text{gO}_2 \text{ m}^{-2} \text{ d}^{-1}$  for the Marsh Point, Buena Vista, Potts Point, and Jones Point stations, respectively. Annual average night-time respiration values were 1.10, 0.77, 0.79, and 0.96  $\text{gO}_2 \text{ m}^{-2} \text{ n}^{-1}$  at these same stations. As is evident in Figure 15, the temporal pattern in primary production is one in which values are generally high during the warm seasons of the year and consistently low during the cool seasons of the year.

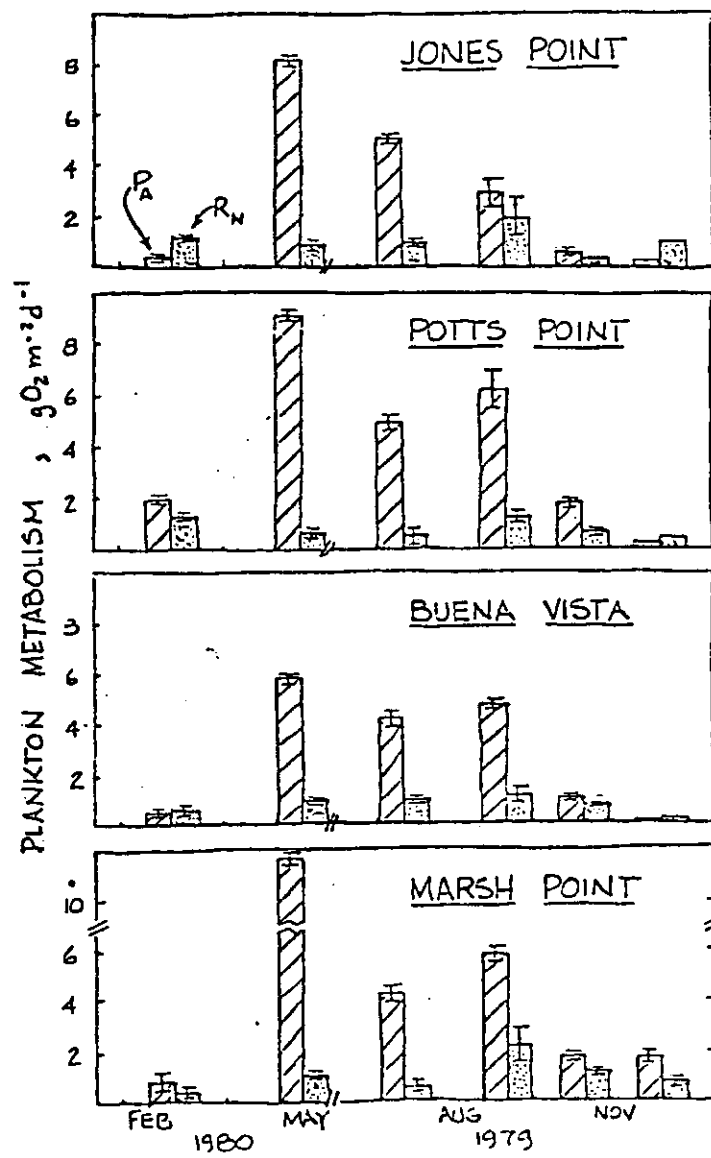


Fig. 15. Seasonal patterns in primary production ( $P_A$ ) and respiration ( $R_N$ ) at four locations in the Patuxent River Estuary, 1979-1980.



Recently, Boynton et al. (1982) reviewed temporal patterns in estuarine phytoplankton production in 63 estuarine systems and found that in most systems high phytoplankton production occurred during warmer seasons of the year.

Somewhat surprisingly, ratios of net production to respiration were consistently equal to or much greater than one throughout the entire study period. In effect, this suggests that a good deal more organic matter was produced in-situ via phytoplankton production than was consumed by heterotrophic processes in the water column. The excess organic matter was, therefore, available for either deposition on the bottom or export to downstream locations in the estuary. If we contrast P:R data collected in this study with that of Cory (1974) we find a distinct contrast wherein at a single station located at the Benedict Bridge (River mile 20) Cory found P:R ratios averaged 0.84 in 1964 and 0.60 in 1969, reflecting a larger respiration in 1969 and perhaps a trend indicating substantial eutrophication. The reasons for these large differences in P:R ratios between these two time periods is not clear, but such differences, if they are real, obviously would have substantial impact on the dissolved oxygen regime in the water column. During the mid to late 1960's there were numerous reports that oxygen in the deeper waters of the Patuxent had declined substantially (Mihurksy and Boynton 1978), an observation consistent with the low P:R ratios observed by Cory. In 1979-1980, we did not observe particularly low dissolved oxygen concentrations in the water column at any of our stations. Data collected in 1980-81 by the Benedict Estuarine Laboratory under contract to Maryland Department of Natural Resources (Water Resources Admin.) indicated only small areas of the estuary

in which dissolved oxygen concentrations were below 2 ppm during the summer. This observation is consistent with the high P:R ratios observed in our study.

D'Elia and Farrell (1982) have analyzed vertical temperature, salinity, and dissolved oxygen data in the Patuxent River estuary, and suggested that strong vertical gradients in dissolved oxygen developed under conditions where there were strong vertical density gradients. Moreover, they indicated that strong density gradients were induced via vertical differences in salinity rather than temperature. In years of low river flow, stratification was weak and reaeration of deep waters via diffusion from the atmosphere was well developed due to frequent mixing of the water column. While this is an attractive mechanism, it appears that more data are needed prior to accepting this as a substantial factor governing oxygen concentrations.

To provide an index of variability concerning  $P_a$ , we have graphically displayed  $P_a$  data for the 1978-79 and 1979-80 periods for two sites for which such data were available (Fig. 16). One obvious characteristic of this diagram is the considerable difference between 1978 and 1979 measurements at the same station. For example, at the Potts Point site,  $P_a$  during May of 1978 was less than  $2 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ , while at the same station in 1979,  $P_a$  exceeded  $8 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ . Similar differences were also evident at the Buena Vista station. Cory (1974) and others have suggested that daily changes in insolation is the prime reason for such variability. In spite of the day-to-day differences noted at identical stations, overall seasonal patterns were very similar between years. Values of  $P_a$  were generally high during the warm periods, and consistently low during the cold seasons of the year. In

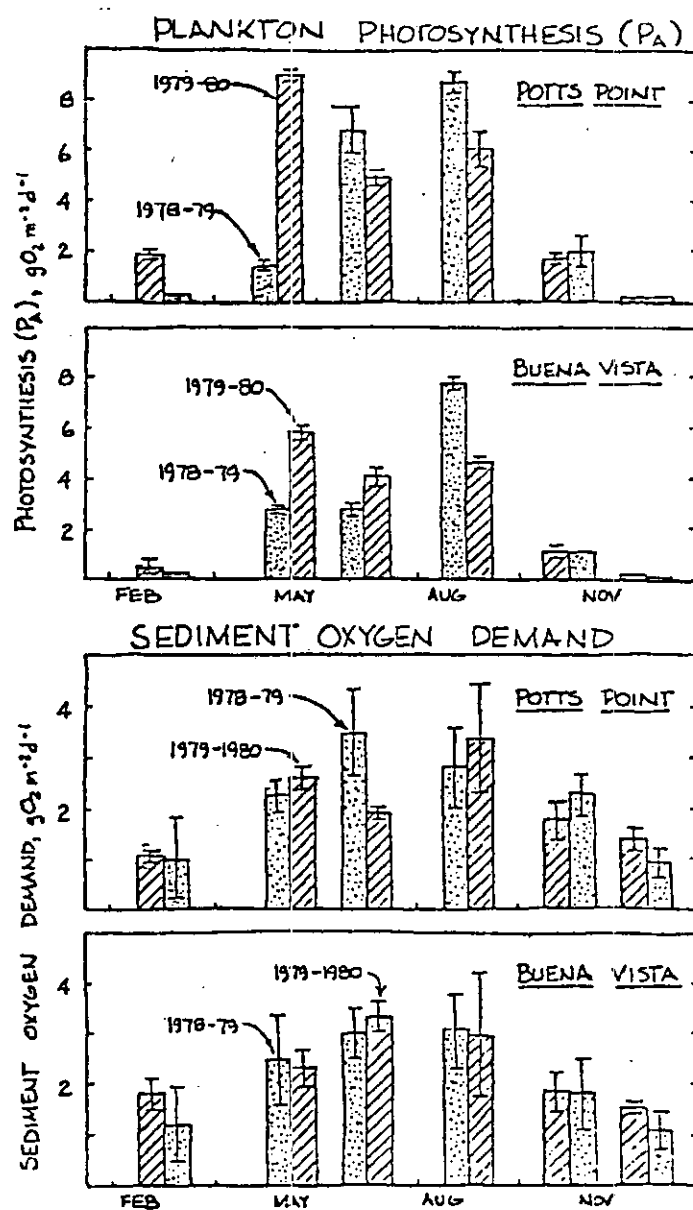


Fig. 16. Comparison of plankton photosynthesis ( $P_A$ ) and SOD for two different years in the Patuxent Estuary.

addition, the time-weighted annual average rate of production was similar between years at the same stations. For example, at the Potts Point station daily average productivity was  $3.4 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$  in 1978 and  $4.1 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$  in 1979. At the Buena Vista station, 1978 and 1979 values were 2.7 and  $2.9 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ , respectively. Thus, while it appears that productivity rates vary substantially from day to day due to changes in such controlling factors as sunlight, overall seasonal patterns emerge in which productivity was high during the warm periods and low during the cold periods, and (at least in the two-year period represented in this data base) annual average rates of productivity were reasonably similar.

In Figure 17, we have prepared a broad review of average daily production rates for a wide variety of estuarine ecosystems, including the Patuxent River estuary. To our knowledge, this represents the most comprehensive review currently available. Average seasonal rates in this diagram range from near 0 to  $2.5 \text{ g C m}^{-2} \text{ d}^{-1}$ . The mean of these rates was about  $0.8 \text{ g C m}^{-2} \text{ d}^{-1}$  ( $292 \text{ g C m}^{-2} \text{ yr}^{-1}$ ), a value substantially higher than the  $100 \text{ g C m}^{-2} \text{ yr}^{-1}$  reported by Ryther (1963) for coastal areas, and quite similar to the  $300 \text{ g C m}^{-2} \text{ yr}^{-1}$  estimated for upwelling systems. Despite the large range in freshwater input, physical morphology, insolation, and other factors, maximum production rates always occurred during the warm periods of the year (June-September), while minimum rates generally occurred during the winter. This pattern has been interpreted to indicate that temperature-regulated metabolism strongly influences nutrient recycling processes and plankton growth rates which are, in turn, important factors in maintaining high photosynthetic rates (Williams 1966; Flemer 1970; Eppley 1972; Nixon 1981).

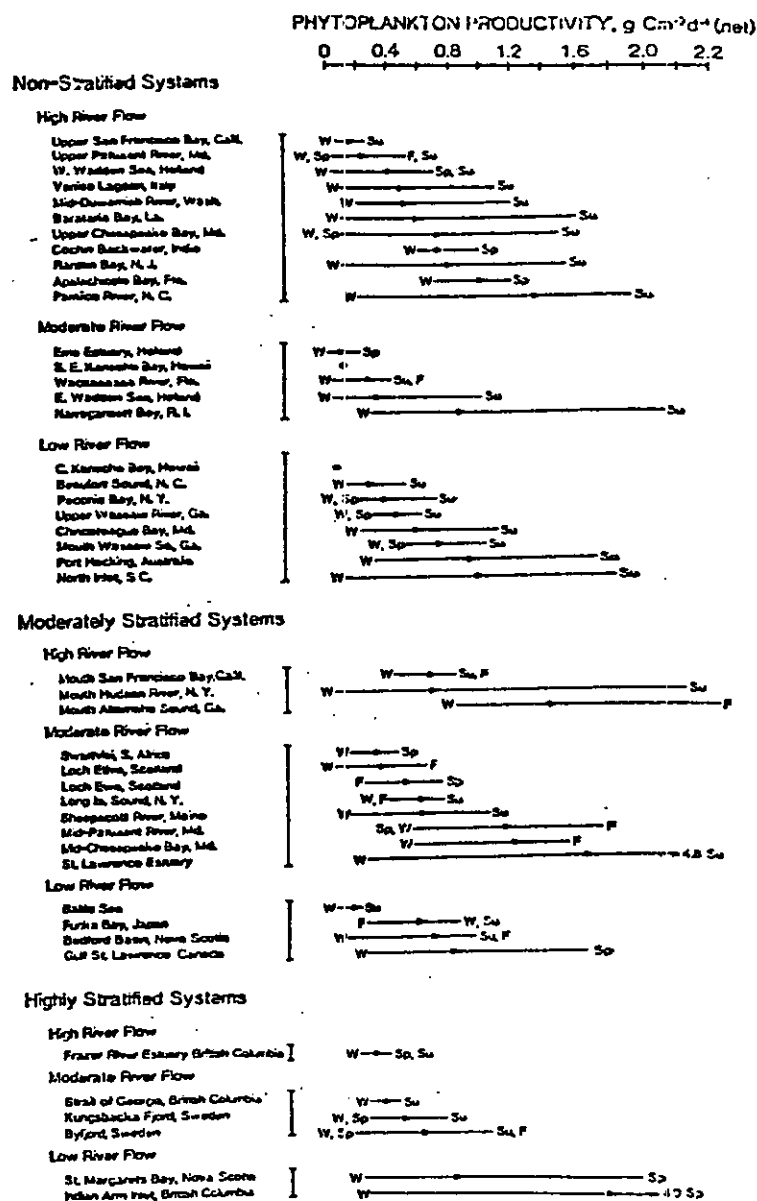


Fig. 17. A summary of average daily phytoplankton productivity in a variety of estuarine ecosystems. Data from Boynton et al. (1982).

Recent calculations concerning the coupling of nutrients and production in Chesapeake Bay indicate that recycling is also dependent on a source of labile organic matter which comes mainly from phytoplankton production, again indicating the tight coupling of heterotrophic and autotrophic processes (Kemp et al. 1982).

While there are considerable differences in the physical structure of the systems shown in Figure 17, the range in production values observed within one group overlaps those of other groups. Annual means for river-dominated estuaries, embayments, lagoons, and fjords were 0.6, 0.4, 0.5, and 0.6  $\text{g C m}^{-2} \text{ d}^{-1}$ , respectively. In short, estuaries with different physical characteristics commonly have comparable rates of primary production, suggesting that there are system-specific biotic and physical mechanisms operative in different estuaries. Values of  $P_a$  in the lower Patuxent estuary were among the highest observed. In fact, higher values were observed only in the Pamlico River, Altamaha Sound, Saint Lawrence, mid-Chesapeake and Indian Arm estuaries. Average daily production estimated from Stross and Stottlemeyer (1965) indicated that average daily rates were about  $1.15 \text{ g C m}^{-2} \text{ d}^{-1}$ . If we combine data from two stations at the lower end of the turbidity maximum for the two years for which data are available, we obtain a time-weighted average daily value of  $P_a$  of about  $1.01 \text{ g C m}^{-2} \text{ d}^{-1}$ , a value very similar to that recorded some years ago.

### Sediment Oxygen Demand

Seasonal patterns ( $\bar{X} \pm S.D.$ ) of SOD are shown in Figure 18 for four stations, and tabular values are given in Appendix 9. Paralleling patterns observed for  $P_a$ , SOD rates were highest during the warm periods, and lower during the cold periods of the year. Considering all stations and sampling times, values of SOD ranged from 4.8 (Potts Point, September 1979) to approximately  $0.0 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$  (Marsh Point, February 1980). Average daily rates were 0.88, 2.05, 2.35, and  $1.31 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$  at the Jones Point, Potts Point, Buena Vista, and Marsh Point stations, respectively.

In earlier work conducted at the Potts Point and Buena Vista stations, linear regression analysis indicated a good correlation between total sediment oxygen demand and water temperature ( $r^2 = 0.68$ ) and a positive intercept ( $0.56 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ ), which was representative of the lowest uptake rate observed during the winter period. A reasonably good correlation was also observed between SOD and primary production ( $r^2 = 0.59$ ). In contrast to this, we found in the present study that SOD data combined for all stations resulted in poor correlations between SOD and a broad variety of environmental parameters (Appendix 7). In this analysis, correlation coefficients ranged from 0.03 to 0.50, and while several were statistically significant, none were reasonable predictors of sediment oxygen demand. One reason that this analysis yielded poor predictors of SOD was that there were substantial differences in the mean rate of SOD at the different stations. When the data from the four stations were treated separately, temperature appeared to be a significant predictor of SOD at several stations. In contrast to this,  $P_a$ ,

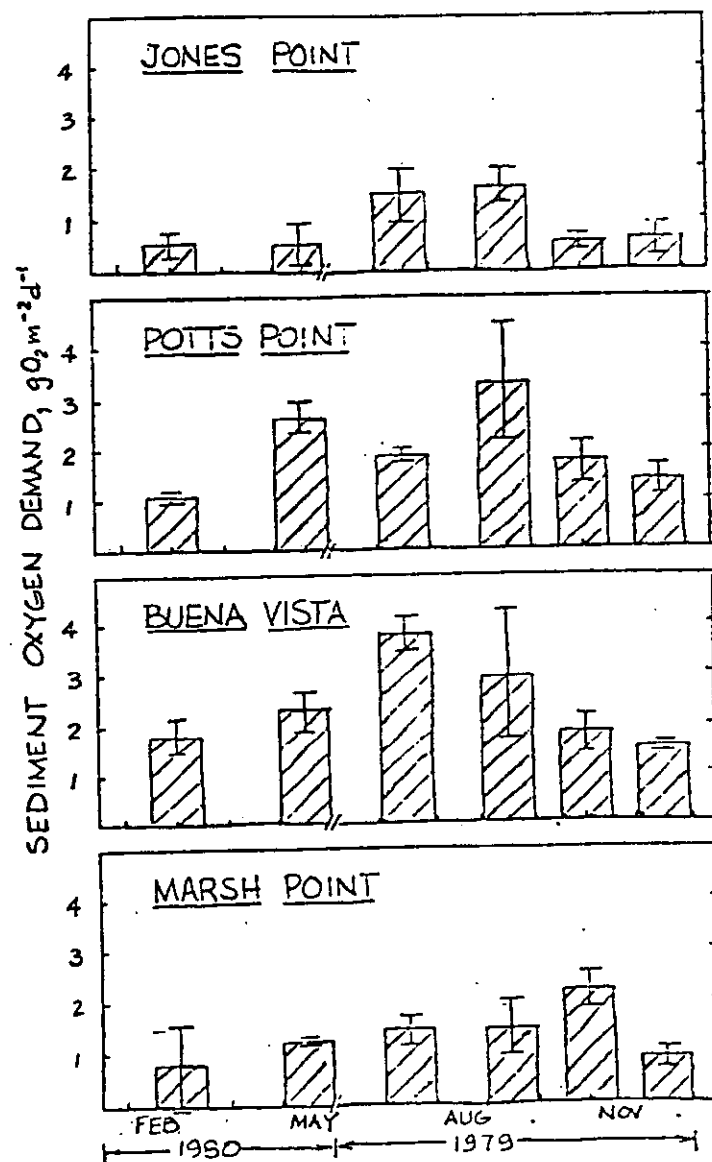


Fig. 18. Average daily values of sediment oxygen demand measured at two stations in the Patuxent River Estuary, 1979-1980.



which was reasonably correlated with SOD in previous work, was significantly related to SOD only at the Potts Point station. SOD was poorly correlated with nutrient concentrations in the water column, salinity, seston, and most sediment nutrient fluxes.

In contrast to the apparent day-to-day variability in rates of  $P_a$ , SOD was quite constant between closely positioned stations (Potts Point and Buena Vista) and between years at these stations (Figure 16). For example, at the Potts Point site, average daily values of SOD were 2.15 and 2.10  $\text{g O}_2 \text{ m}^{-2} \text{ d}^{-1}$  in 1978 and 1979, respectively. Similarly, at the Buena Vista station, 1978 and 1979 values were 2.10 and 2.27  $\text{g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ , respectively. The stability in these rates may in part be due to the active cycle of organic matter deposition and resuspension which apparently reaches a maximum in this zone of the estuary. With high daily rates of resuspension and deposition, heterotrophic organisms in the sediments may be exposed to a more constant supply of organic matter than if they were strictly dependent on obtaining some fraction of the daily water column production which is known to vary considerably over daily time intervals.

As indicated in Figure 18, a pattern in SOD along the longitudinal axis of the estuary emerged, wherein the highest rates occurred in the turbidity maximum portion of the estuary with lower values generally observed both upstream and downstream of this section of the estuary. The high flux rates observed in the middle section of the estuary may well be related to both the seasonal supply of organic matter to the benthos and to the physical dynamics of the sediment/water interface. With regard to the latter, results of

sediment trap studies suggest very high rates of sedimentation and resuspension processes which are known to enhance SOD. Davies (1975) and Boynton et al. (1981) have shown that increased water velocities and resuspension of surficial sediments enhances sediment oxygen demand through such mechanisms as enhancing the availability of organic matter to heterotrophs, more quickly removing waste products, and exposing reduced compounds to oxygenated water.

In Figure 19, we provided a comparison of sediment oxygen demand in several zones of the Patuxent River estuary with rates observed in a variety of other estuarine and coastal ecosystems. Clearly, average daily SOD in three areas of the Patuxent River estuary are substantial relative to those observed in a variety of other ecosystems. Moreover, the mean rate observed in the turbidity maximum zone is among the highest yet recorded in a variety of ecosystems. The particularly high value observed in the turbidity maximum may result from the combination of factors previously discussed.

Rates of sediment oxygen demand were analyzed with respect to statistical relationships to such variables as deposition of organic matter, the amount of organic carbon content of sediments and the P:R ratio of plankton metabolism in the water column. All of the above are indices of the organic matter available to the sediments. Linear regressions analyses of the data in Table 11 revealed a strong relationship between annual carbon deposition and summer rates of sediment oxygen demand ( $r^2 = 0.87$ ;  $P > 0.05$ ), but regressions were non-significant for the remaining variables. Our interpretation of this

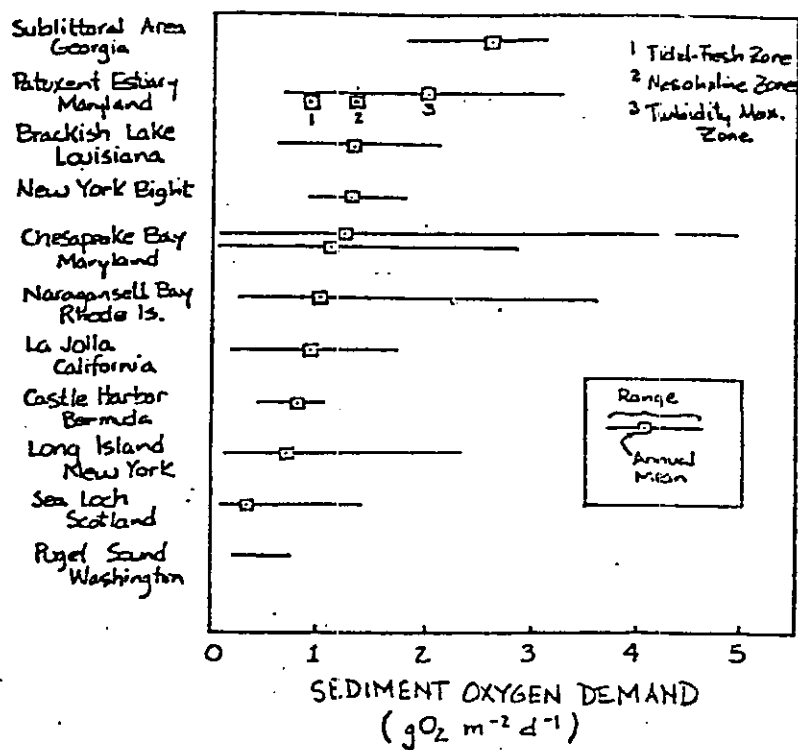


Fig. 19. A comparison of SOD (annual mean rates and range) in a variety of estuarine and coastal ecosystems.

TABLE 11. ANNUAL MEAN RATES OF SEDIMENT OXYGEN DEMAND IN RELATION TO SEDIMENT DEPOSITION RATE AND % ORGANIC CARBON IN SEDIMENTS.

STATION	SOD* ( $\text{gO}_2\text{m}^{-2}\text{d}^{-1}$ )	Carbon# Deposition ( $\text{Kg m}^{-2}\text{y}^{-1}$ )			Sediment Organic-C <sup>+</sup> (%)	Plankton P:R++
		I	II	III		
St. Leonards Cr.	---	0.28	0.29	0.38	---	---
Marsh Point	1.5	0.26	0.40	0.48	3.0	7.3
Buena Vista	3.3	1.08 (1.51)	0.45 (0.71)	0.36 (0.71)**	3.5	5.9
Potts Point	2.6	1.32	0.42	0.54	4.2	14.4
Jones Point	1.6	0.47	0.28	0.33	4.0	3.8

\*Mean summer rates.

\*\*Mean of deep and shallow stations for sediment trap arrays.

+May values only

++May - October values.

#Net sedimentation of carbon calculated from sediment trap data using three different sets of assumptions.

- I. The cup containing the minimum amount of sediment on each deployment was used to calculate net deposition; the % carbon associated with that cup was used to estimate net carbon deposition.
- II. As in (I) but extreme values were omitted.
- III. An average value was generated from 2-3 near-surface collection cups and used to calculate net sedimentation. Extreme values were not used.

analysis is that SOD is most proportional to the amount of organic matter reaching the sediment surface, a result consistent with the analyses reported by Hargrave (1973) and Nixon (1979), both of whom included a number of different estuarine systems in their studies.

It may be that the percent of organic matter in sediments represents more the residual amount of organic matter remaining after labile material was consumed in heterotrophic processes. This is particularly true in the data we used because the sediment organic matter content was calculated from a bulk sample which was composed of material in the top 10-20 cm of the sediment. It may be that a significant regression might have emerged if it had been possible to sample only the top 1 cm or less, and hence capture within the sample labile organic matter that had very recently been deposited on the sediment surface and not yet metabolized. The P:R ratios reported in Table 11 were calculated based upon single-day measurements, and may represent too short a time-span to be representative of the general picture of organic matter residuals remaining in the water column. An alternative explanation for the lack of a relationship here is that such water column characteristics as P:R may not reflect the amount of material available for deposition at the place in the estuary where it was measured. For example, it is possible that production generated at one point is deposited a considerable distance either upstream, or more probably downstream. In any case, consistent with previous research findings, we find a strong relationship between sediment oxygen demand and the rate of deposition of organic matter to the bottom measured directly from sediment trap deployments.

### Dissolved Oxygen Characterization

We have developed surface and bottom time-space plots of dissolved oxygen (DO) concentration in the Patuxent River estuary, as well as diel plots of temperature, salinity, surface and bottom dissolved oxygen concentration, percent saturation, mean DO and tidal height for several locations within the estuary (data from Benedict Estuarine Laboratory). Our purpose here was to utilize this large water quality data base to characterize oxygen concentrations throughout the estuary towards developing oxygen budgets for this estuarine system. In addition, we report here new data concerning oxygen fluxes across the air-water interface.

Surface and bottom dissolved oxygen concentrations plotted in the time-space domain are given in Figure 20. Oxygen concentrations were high in the winter and low in the summer in both surface and bottom waters reflecting in part the dependence of oxygen solubility on temperature. Oxygen concentrations ranged from about 5 - 13 mg-l<sup>-1</sup> in surface waters and from about 1-11 mg-l<sup>-1</sup> in bottom waters. The time and space plots indicate considerable variability in dissolved oxygen concentration over time spans of one to several weeks and longitudinal distances are from 10 to 20 km. This observation is of particular importance in developing oxygen budgets for the estuary because tidal excursion distances approach the inter-oxygen patch distance and, hence, stationary monitoring of oxygen within the estuary could well be seriously confounded.

Periods of low dissolved oxygen occurred primarily during July and August in this data set and extended from 10-80 Km above the mouth of the estuary.

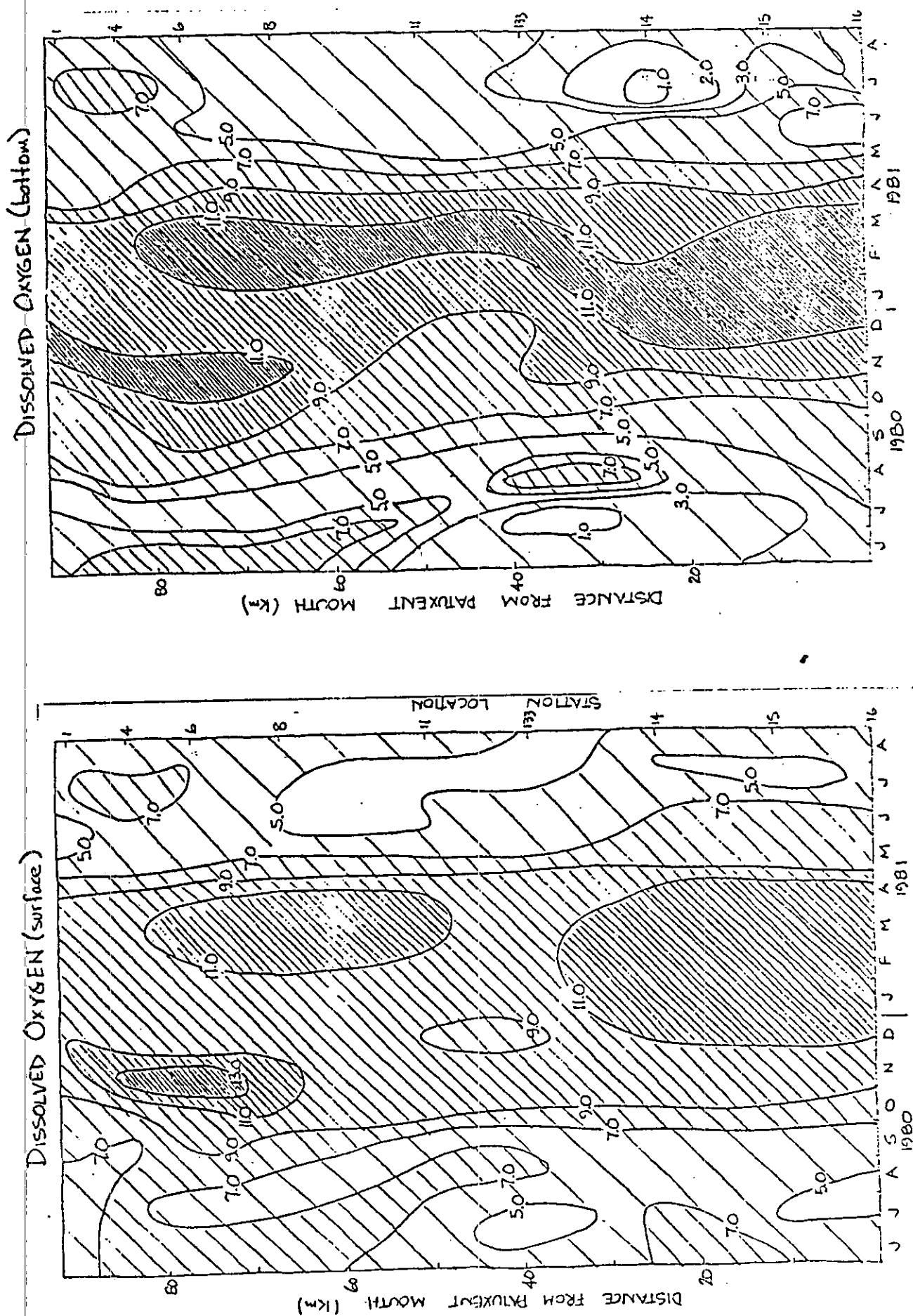


Fig. 20. Dissolved oxygen concentrations in surface (A) and bottom (B) waters of the Patuxent River Estuary, 1980-1981. Data from Benedict Estuarine Laboratory. The Chalk Point Site is located at river km 45.

The epicenters of low dissolved oxygen concentration were more restricted during both summers and were located between 13-40 Km from the mouth of the estuary. In comparison to earlier records of dissolved oxygen concentration, the values reported in this data base were somewhat higher. Mihursky and Boynton (1978) suggested that considerable year-to-year changes occur in many water quality parameters, dissolved oxygen among them. It may be that the dissolved oxygen picture captured in the 1980-81 water quality survey represents a "good water quality year" as opposed to a bad year, but since monitoring programs do not run continually in this estuary, it is virtually impossible to address such important trend-related questions.

In Figure 21, we have plotted diel patterns of temperature, salinity, dissolved oxygen, percent saturation and tidal height for three locations in the Patuxent River estuary (Sheridan Point, Potts Point and Nottingham) on July 24-25, 1980. In terms of DO, the general pattern appears to be fairly strong, and involves an increase in dissolved oxygen concentration during the daytime and a decline during the night time, reflecting the predominance of photosynthetic and respiratory processes during the diel period. In general, during the daylight, oxygen concentrations exceeded 100% saturation at prevailing temperatures and salinities, while at night waters were undersaturated with respect to oxygen.

However, the short-term dynamics were clearly more complex as indicated by brief periods in which oxygen concentration increased at night in both surface and bottom waters, a process which obviously is not related to photosynthetic activity. In fact, such excursions in oxygen are probably the



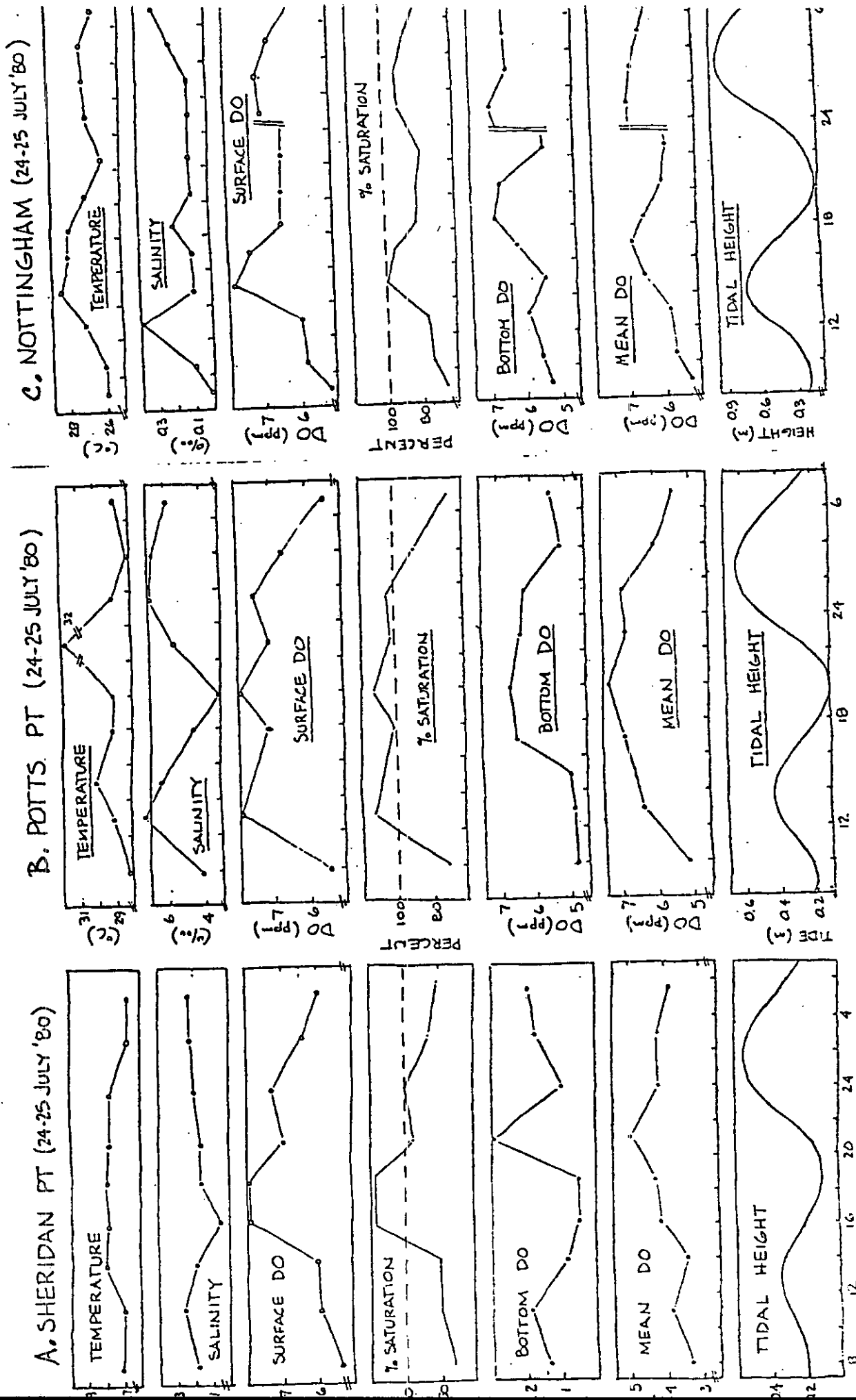


Fig. 21. Diel values of temperature, salinity, DO, percent saturation and tidal height for selected stations in the Patuxent River Estuary. (A) Sheridan Point, (B) Potts Point, (C) Nottingham.

result of water with a different oxygen history advecting into the zone where oxygen measurements are being made. Hence, it does not appear to be possible to use the single station method of measuring community metabolism following the technique of Odum and Hoskin (1957).

In Figure 22, measurements of oxygen flux across the air/water interface are plotted against current speeds at the time of measurement. Tabular data are given in Appendix 10. Gas transfer coefficient values ranged from about 1 - 3.5  $\text{g O}_2 \text{ m}^{-2} \text{ d}^{-1} \text{ atmosphere}^{-1}$  over a current velocity range of from 10 - 40 cm/sec, and in water of 1-6 m. As plotted here, there appears to be a weak relationship between the magnitude of the gas transfer coefficient and water velocity, but clearly additional analysis, probably utilizing other independent variables, will be necessary before stronger relationships emerge.

We suggest that it would be possible to construct dissolved oxygen budgets for distinct portions of the estuary using data such as presented here. For example, the diel change in the total amount of oxygen in the water column is nicely described by the data presented in Figure 21. Oxygen demand by the sediments and oxygen demand in the water column could be estimated using independent measurements of plankton respiration and benthic respiration, respectively. Based on the percent saturation of the water column and oxygen gas transfer coefficients, the magnitude of the oxygen flux across the air/water interface might also be approximated. Finally, oxygen production in the water column could be estimated from bottle measurements of plankton metabolism, such as those provided in Figure 14. In addition, vertical transfer of oxygen could be calculated based on observed oxygen

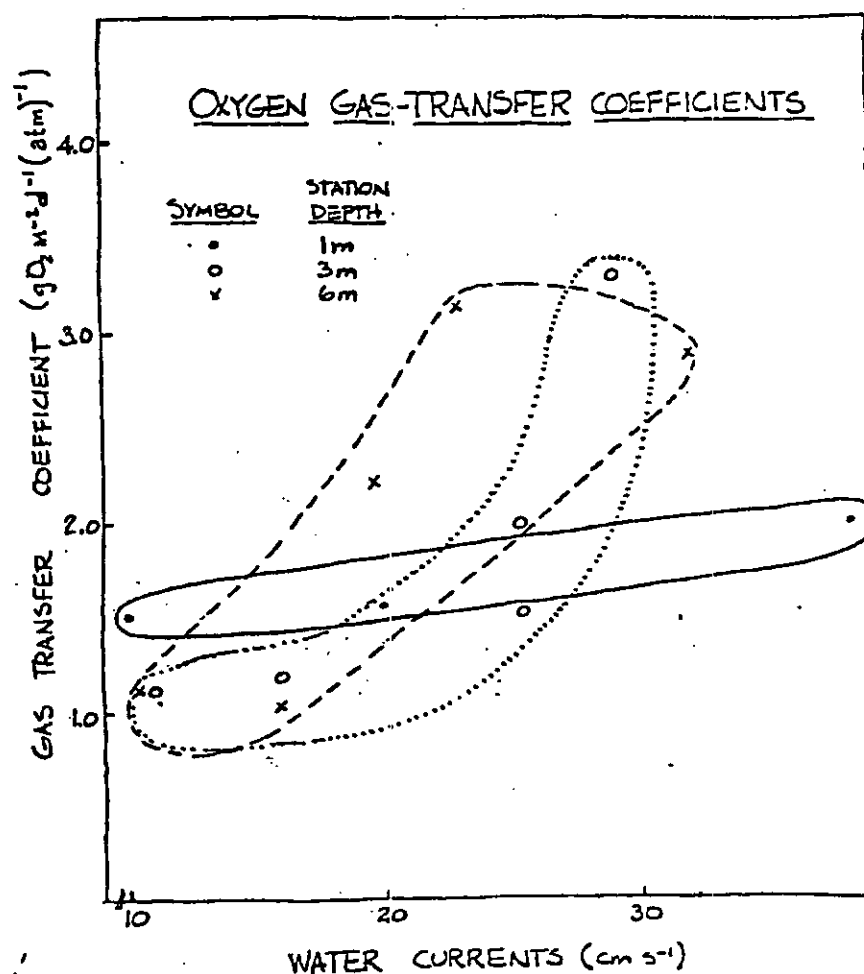


Fig. 22. Oxygen transfer coefficients measured in waters of different depths and plotted against surface current speeds at the time of measurement.

gradients, and vertical diffusivity coefficients could be selected on the basis of the strength of vertical stratification. The last term necessary in constructing an oxygen budget would be estimating the horizontal, advective component, and in this case could be solved by difference. This approach is similar to that used by Kemp and Boynton (1980) who developed oxygen budgets for two locations in the open Chesapeake Bay. Such an approach allows one to distinguish between physical and biological influences on the oxygen budget, and hence has utility concerning the management of oxygen in an estuary vis-a-vis providing a quantitative basis from which one can decide which term in the oxygen budget would produce significant effects concerning oxygen concentration and be amenable to available management techniques.

## SESTON CHARACTERISTICS

### Seston and Chlorophyll a

Seston in the Patuxent River estuary was only sparsely characterized in our study due to the fact that longitudinal seston distributions were measured on only four occasions. To more fully characterize seston distributions, we utilized the water quality data generated by the Benedict Estuarine Laboratory for the Maryland Water Resources Administration. Surface seston concentrations are displayed in the time/space format in Figure 23. During the 1980-1981 period, seston concentrations were reasonably low, ( $<40 \text{ mg}^{-1}\text{L}$ ) in the lower 24 miles of the estuary.

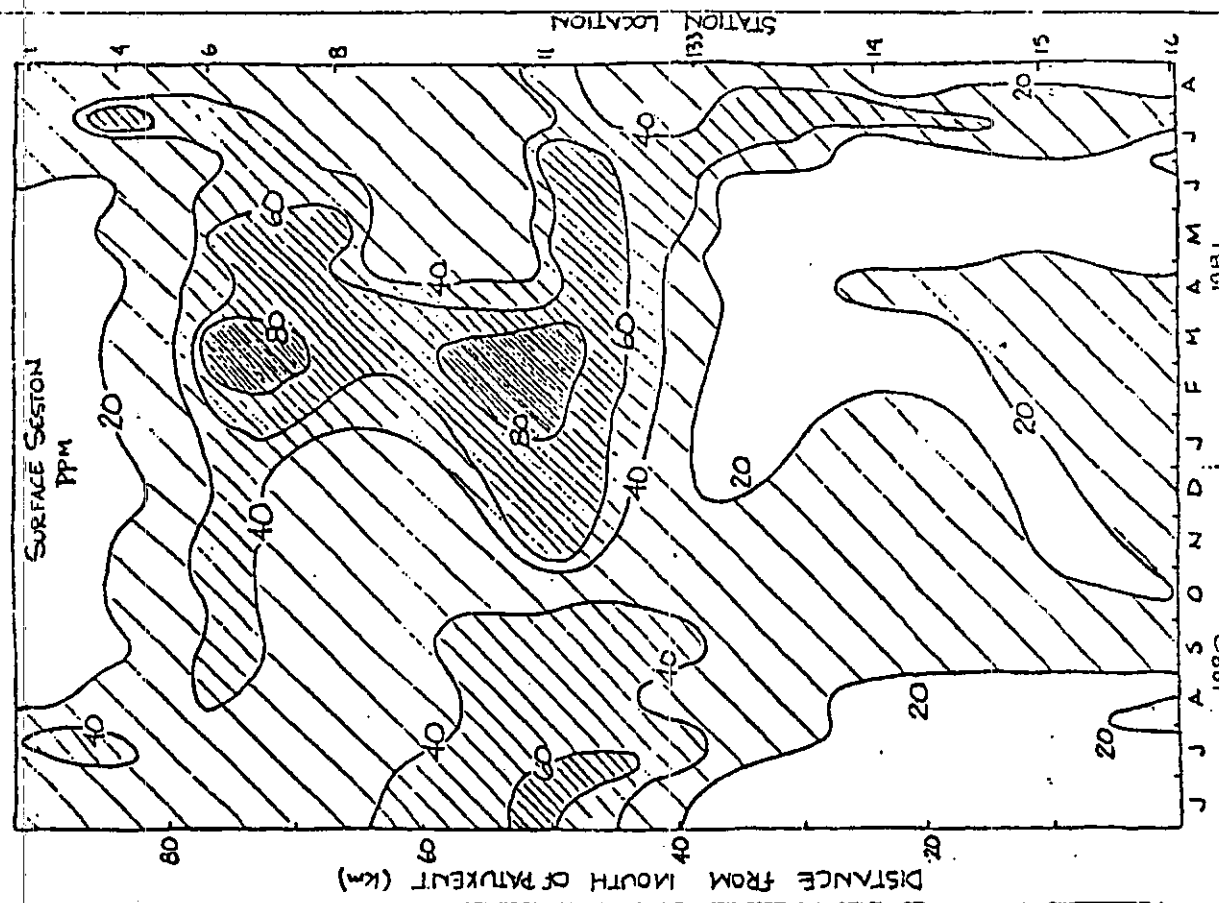
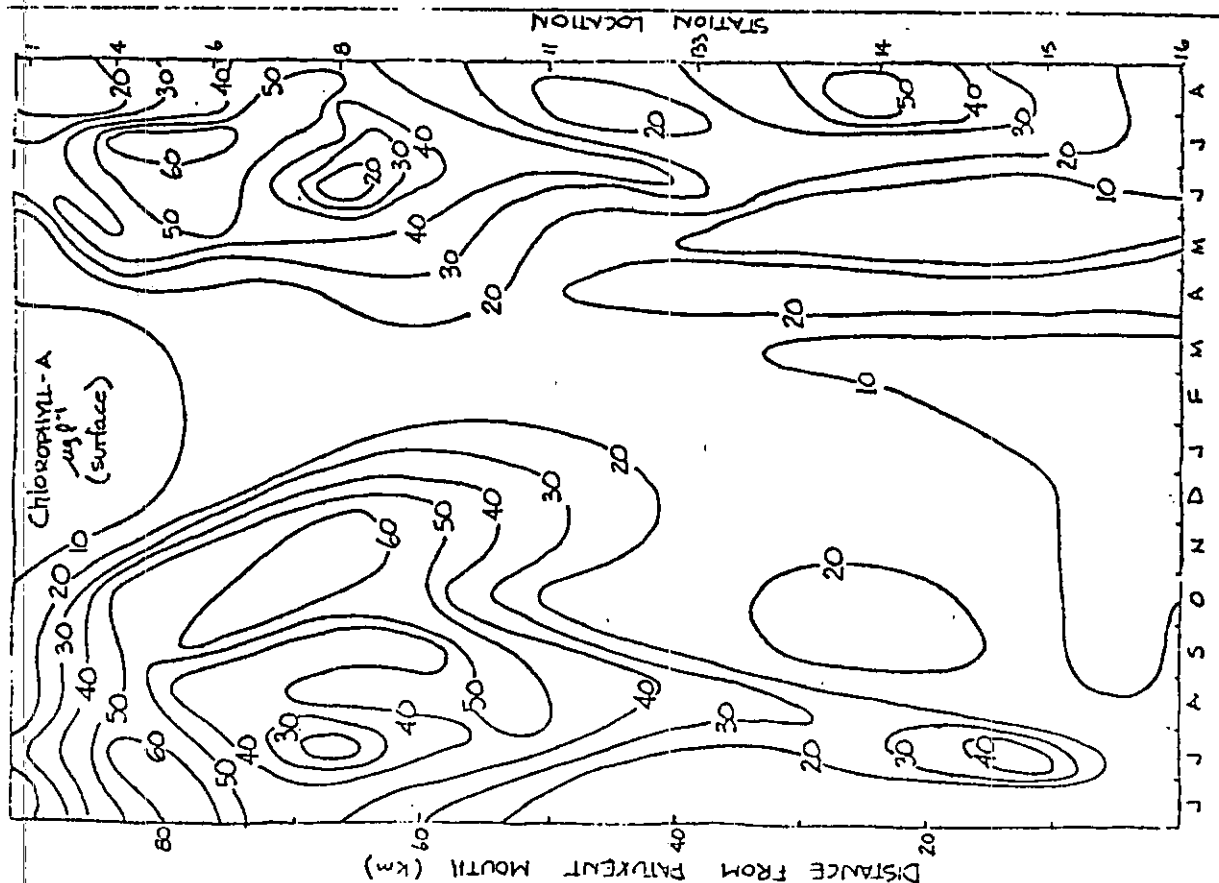


Fig. 23. Seston and chlorophyll a distribution in surface waters of the Patuxent River Estuary, 1980-81. The Chalk Point Site is located near river km 45.

In both years, there was a clearly defined turbidity maximum zone in the upper estuary. In the latter half of 1980, this zone was located between river km 40-75, where concentrations exceeded  $60 \text{ mg}^{-1}\text{L}$ . In 1981, the downstream edge of the turbidity maximum was at km 40, but extended considerably farther upstream (km 80), was more intensely developed (concentrations  $> 80 \text{ mg}^{-1}\text{L}$ ), and a double maximum developed at river km 50 and 75. Unfortunately, only one winter period is included in this data base, and it appears that in the winter, the turbidity maximum is best developed. Hence, comparisons between 1980 and 1981 are not possible. The characteristics described here are similar to those reported by Keefe et al. (1976) who found the turbidity maximum to extend over similar portions of the river, and also reported finding two turbidity maxima at times. The Chalk Point site was near the lower end of the turbidity maximum in both years.

A second component of the seston, chlorophyll a, is plotted in time-space format in Figure 23. In general, chlorophyll concentrations were high in summer and low in winter, a pattern which has been observed repeatedly in the Patuxent as well as many other coastal and estuarine systems (Boynton et al. 1982). Highest chlorophyll concentrations were generally observed above river km 50, but in several occasions in both 1980 and 1981, there were indications of reasonably intense phytoplankton blooms at downriver locations. It is interesting to note that seston and chlorophyll concentrations do not overlap particularly well in the time-space format. Seston concentrations were generally near maxima in the winter, while chlorophyll concentrations are much higher in the summer. We might expect, based on seasonally changing characteristics of seston, that there would be differential effects on light